

VI. FACTORS AFFECTING THE ABILITY TO SURVIVE AND REPRODUCE

Climatic Variation

California experiences wide variation in climatic and hydrologic conditions. Various climatic phenomena including severe storms, drought, seasonal cycles, El Niño/La Niña events, decadal events, and regime shifts can alter the physical, chemical, and biological aquatic environment (Parrish and Tegner 2001). These changes can, in turn, play a major role in the life cycle, productivity, and persistence of coho salmon populations. Climatic fluctuation can cause extreme conditions that can be catastrophic. Coho salmon evolved with, and have persisted in the face of, extreme variability in habitat conditions caused by these natural phenomena. However, catastrophic conditions combined with low population numbers, habitat fragmentation, anthropogenic impacts, and habitat destruction or loss can cause an unrecoverable decline of a given population or species (Moyle et al. 1995).

Drought

In California, coho salmon populations exist in many coastal streams where stream closures occur due to sand bar formation at their mouths, created through wave action and low summer flows. Coho salmon are able to identify their natal stream by the seepage of fresh water entering the ocean through the bars, but they can't enter the streams until fall or winter rains increase flows sufficient to breach the sand bars. Shapovalov and Taft (1954) found that streams south of San Francisco may not be passable until as late as March. When this happens, a large portion of the run may enter the stream over a short period. As much as 70% of the total escapement may enter the stream from the ocean in as little as a few days (Sandercock 1991). During prolonged droughts, sand bars may never open in a given season and spawners may not be able to enter those streams (Anderson 1995).

Evidence from tree growth rings and other sources suggest that droughts persisting for decades have occurred in California in the past 1,000 years (NMFS 1997; Dettinger 2001). During these periods, stream flows may decrease, constricting or separating available habitat (Spence et al. 1996). Reduced flows can cause increases in water temperature, resulting in increased heat stress to fish and thermal barriers to migration. Anderson (1995) noted that desiccation of rearing and holding areas could eliminate year-classes or entire populations. Drought conditions along the Pacific coast in recent years may have depressed freshwater salmon production (Myers et al. 1998). Droughts can have limited benefits to fish including stabilization of stream features by allowing encroachment of vegetation into the active stream channel.

Flooding

Flooding is usually caused by heavy precipitation in a given watershed over a relatively short period, though smaller storm events can cause flood conditions in urban watersheds due to increased surface runoff (Booth 1991). High flows associated with floods can cause complete loss of eggs and larvae as they are scoured from the gravel or buried in the sediment (Sandercock 1991; NMFS 1998). Juveniles, smolts, and adults can also be affected by flooding. Smolts and juveniles can be transported and stranded on the floodplain or washed downstream to poor habitat, out to sea prematurely, or into isolated side channels and off-channel pools. Adults can be affected by peak flows influencing them to move into isolated channels and pools, or

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preventing migration through excessive water velocities. Streams can be drastically modified by erosion and siltation in large flood flows almost to the extent of causing uniformity in the stream bed (Spence et al. 1996). After major floods, streams can take years to recover pre-flood equilibrium conditions. Flooding is generally not as devastating to salmon in the more morphologically complex streams, because protection is afforded to the fish by the variety of natural in-stream structures (LWD, boulders, root wads, etc.), stream channel types (pools, riffles, side channels, etc.), and a established riparian area (Spence et al. 1996). Some of the beneficial effects of flooding are: cleaning and scouring of gravels, transporting sediment to the flood plain, moving and rearranging LWD, recharging flood plain aquifers (Spence et al. 1996), allowing salmonids greater access to a wider range of food sources (Pert 1993), and maintaining the active channel.

Changes in Ocean Conditions

Salmon abundance is known to be extremely variable. Bisson et al. (1997) estimated that salmon population size can experience year-to-year variation from 40-70%. Numerous sources have concluded that variation of salmon run-size and spawner escapement are strongly affected by changes in the marine and freshwater environment (Pearcy 1992; Beamish and Bouillon 1993; Lawson 1993). Relatively short term El Niño events (occurring at approximately three- to four-year intervals) and longer decadal to inter-decadal shifts are both known to affect marine organisms including salmon (Parrish and Tegner 2001).

El Niño and La Niña events have had major impacts on the primary and secondary productivity of the North American Pacific coast through changes in the thermal regime which limit upwelling and nutrient replacement (Brown et al. 1994). The term El Niño describes complex and large scale changes in the atmospheric pressure system, trade winds, and sea surface temperatures that occur over the entire tropical Pacific (Parrish and Tegner 2001) that can affect salmon production (Appendix B2). La Niña oceanic conditions are characterized by unusually cold ocean temperatures, relative to El Niño conditions, in the Equatorial Pacific (NOAA 2001). Marine productivity depends on atmospheric and oceanic circulation which affects the abundance of salmonids and other fishes. These shifts in wind, upwelling, and ocean currents have caused declines in ocean survival and decreases in size of coho and chinook salmon (Johnson 1988; Spence et al. 1996; Tschapinski 1999).

Holtby et al. (1990) indicated that large numbers of predatory fish move northward in these conditions, possibly causing a major impact on the smaller first-year ocean coho salmon. The 1983 El Niño event off the Pacific coast of North America resulted in increased adult mortality and decreased average size for Oregon's coho and chinook salmon (Johnson 1988). That paper also noted that coho salmon entering the ocean in spring of 1983 survived poorly, resulting in low adult returns in 1984, and that the average weight of coho and chinook salmon landed in 1983 by Oregon's commercial troll fishery was the lowest ever recorded. Another component of El Niño is the flooding conditions it often brings to inland areas. U.S. Geological Survey streamflow hydrographs show that, in California, El Niño years are more likely to exhibit high flows or flood conditions than normal or La Niña years (Cayan et al. 1997).

In a La Niña event, the West Wind Drift is diverted south towards California. Typically, this happens when a weak low pressure forms south of Alaska (NOAA 2001). Generally, ocean conditions are cooler and possibly more favorable to salmonids during La Niña events; however,

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inland conditions caused by this phenomenon can affect survival of juveniles. Colder weather conditions are generally descriptive of La Niña events, but, depending on its severity, there could be associated drought or flooding (Null and Monteverdi 1999). Data from the San Francisco monitoring station indicated that during the 1975 La Niña, total seasonal rainfall was well below 50% of normal, whereas in the 1973 event, it was over 130% of normal. Null (2001) noted that high total seasonal rainfall does not necessarily denote major flooding conditions, but rather it is the timing of the storms within the season that determine the impact. The storms that create the most flood damage are those that occur as high intensity-short duration precipitation events.

Spence et al. (1996) noted that any changes in surface currents and upwelling strength will influence temperature, salinity, and nutrients, thereby affecting the abundance of food available to juvenile salmonids, the number and distribution of predators and competitors, and the transport of smolts entering the ocean (along-shore versus off-shore). Recent evidence suggests that when ocean conditions are poor for salmonids in the Pacific Northwest, conditions are favorable to Alaskan stocks, and vice-versa.

Recent studies have shown that longer time-scale (decadal to multi-decadal) changes have affected, and are currently affecting, marine organisms in California, including coho salmon (Parrish and Tegner 2001). Hare and Francis (1995), Beamish et al. (1997), Beamish et al. (1999), Pearcy (1992), and Lawson (1993), among others, describe recent and historical correlations between large-scale physical ocean changes, ocean productivity, and Pacific salmon abundance. Several recent studies have related ocean conditions specifically to coho salmon production (Cole 2000), ocean survival (Ryding and Skalski 1999; Koslow et al. 2002), and spatial and temporal patterns of survival and body size (Hobday and Boehlert 2001).

The Pacific Decadal Oscillation, indicated by changes in winter-time North Pacific atmospheric circulation, is associated with regime shifts in the subarctic and California Current ecosystems. These shifts are out of phase with one another, such that when conditions are good in the subarctic they are poor in the California Current, and vice-versa (Koslow et al. 2002). Cole (2000, based on results in Francis and Hare 1994, Gargett 1997, and Mantua et al. 1997) stated that warm conditions in the northeast Pacific generally favor Alaskan salmon stocks, whereas cooler conditions appear to favor stocks south of British Columbia. A major regime shift that occurred in 1976/77 resulted in warmer surface waters from Mexico to Alaska. Resultant changes in ocean production negatively affected most salmon stocks south of British Columbia, including coho (Myers et al. 1998).

Conclusions

Long-term trends in the ability of freshwater environments to support salmonids may not be evident during periods of favorable oceanic conditions (Spence et al. 1996). Stated differently, favorable marine conditions can mask the effects of freshwater environmental degradation on salmonid populations. The cyclic nature of marine productivity as outlined by Lawson (1993) shows how it can mask the decline of a salmonid population (Figure 23). The conceptual model he presents combines the effects of oceanic cycles and freshwater habitat degradation. As the habitat degrades, the salmon populations do not decline in a linear fashion. Instead, due to the long-term cycles of productivity in the marine environment, the downward trend can be masked by higher escapement due to more favorable oceanic condition. In periods when unfavorable ocean conditions coincide with freshwater habitat degradation, the

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consequences are more evident, and the possibility of local extinction becomes greater for salmonid populations in degraded watersheds.

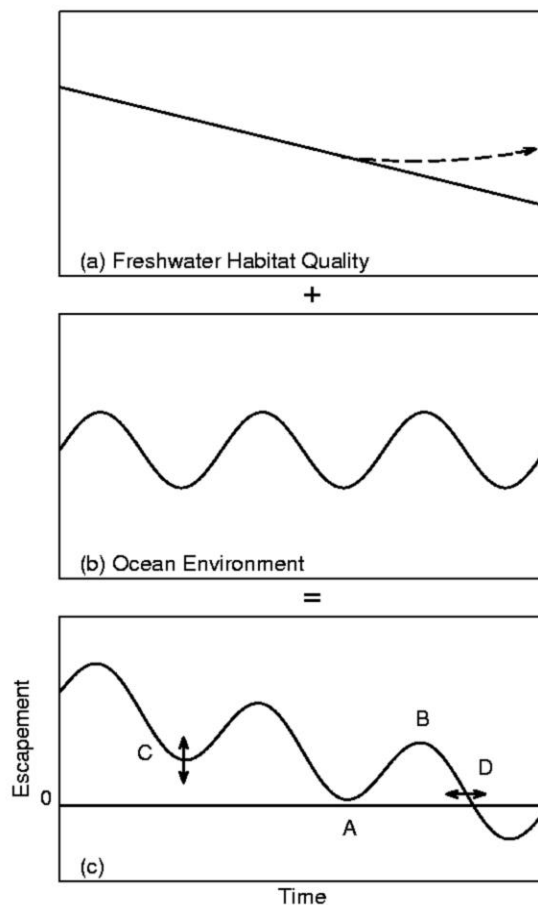


Figure 23. Conceptual model of effects of declining habitat quality and cyclic changes in ocean productivity on the abundance of coastal natural salmon. Top chart shows trajectory of habitat quality over time, with the dotted line representing possible effects of habitat restoration projects. Middle chart shows the cyclic nature of ocean productivity. Bottom chart shows the sum of top two panels where letters represent the following: A = current situation, B = situation in the future, C = change in escapement due to increasing or decreasing harvest, and D = change in time of extinction due to increasing or decreasing harvest (from Lawson 1993).

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During periods of favorable environmental conditions, reproduction by individuals that colonize marginal patches might contribute to greater overall abundance and could buffer the effects of environmental variation when conditions worsen (McElhaney et al. 2000). However, a fragmented species consisting of small populations separated by large geographical distances and a high rate of loss of available habitat patches is vulnerable to catastrophic loss due to environmental fluctuations because each population is isolated and may be too small to be viable (McElhaney et al. 2000). In general, small populations with limited or fragmented distributions are more vulnerable to decline or extinction due to stochastic processes, cyclic events, and extreme climatic variation than larger populations with broad distributions. The National Research Council report on salmon in the Pacific Northwest (NRC 1996) recommended that maintenance of metapopulation structure with good geographic distribution should be a high management priority to ensure long-term perpetuation of salmon populations. Productivity of small populations may also be low due to depensation, which tends to prevent small populations from quick recovery from catastrophic events (McElhaney et al. 2000). Clearly some, if not many, California coho salmon populations are currently small and fragmented.

Shifts in salmon abundance due to climatic variation are known to be large and sudden (Beamish et al. 1999). These changes may cause increases in local extinction rates. However, the Department believes that climatic variation (e.g. regime shifts) resulting in changes in ocean productivity and salmon abundance is generally not likely to threaten California coho salmon with extinction unless population sizes are low, distribution is limited, and metapopulation structure is fragmented. If these features are already compromised for other reasons, then natural shifts in abundance might increase local extinction rates and decrease potential for colonization enough to cause widespread extinction.

Changing ocean conditions, extreme climatic conditions, and natural variation can strongly impact salmon populations. However, salmon populations have not, until the past century or so, experienced these conditions in conjunction with the widespread human related degradation of their spawning streams (Brown et al. 1994; Anderson 1995). Anthropogenic factors can interact with natural variation to increase the frequency of catastrophic conditions (Bisson et al. 1997). Lawson (1993) concluded that the risks associated with poor climatic conditions may be exacerbated by human influence. Salmon evolved in a variable environment and are well suited to coping with it (Bisson et al. 1997). However, declines in population size, cohort loss, and population fragmentation likely reduce the ability of natural populations to respond to extremes of natural environmental variation. This may be especially true for species at the edge of their range like California coho salmon. Small populations can be forced to extinction by environmental variation when survival or productivity are frequently reduced over a long period of time (McElhaney et al. 2000)

Disease

Disease is a result of a complex interaction between the host, disease agents, and the environment. Natural populations of salmon have co-evolved with diseases that are endemic to the Pacific Northwest and have developed a level of resistance to these pathogens. Their resistance to different pathogens makes them unique and the most fit for that environment. For example, in the Columbia River system where ceratomyxosis, caused by the protozoan *Ceratomyxa shasta*, is prevalent, native salmonids have developed resistance to the pathogen and are not as severely affected as exotic stocks introduced by the hatcheries (Stoskopf 1993).

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Generally, diseases and parasites do not cause significant mortality in native coho salmon stocks in natural habitats (Bryant 1994), and Shapovalov and Taft (1954) reported this to be the case in Waddell Creek during the 1930s and 1940s. Understanding mortality caused by pathogens in the wild is limited by the difficulty in determining the proximate and ultimate causes of death (e.g. when fish weakened by disease are consumed by predators). Currently, there is insufficient data from which to draw meaningful conclusions about the importance of disease in regulating wild populations (see comments, Appendix B2).

Coho salmon are susceptible to an array of bacterial, viral, parasitic, and fungal diseases found in salmonids of the Pacific Northwest. Symptomatic conditions appear when fish are stressed by high water temperatures, crowding, environmental contaminants, or decreased oxygen (Warren 1991). Diseases affect various life stages differently. Some of the diseases and disease agents in California that can cause significant losses in adult salmonids include: bacterial kidney disease (BKD) (*Renibacterium salmoninarum*), furunculosis (*Aeromonas salmonicida*), columnaris (*Flexibacter columnaris*), pseudomonas/aeromonas, and ichthyophthirius or “ich” (*Ichthyophthirius multifiliis*) (William Cox pers. comm.). The diseases that are known to cause significant losses in juvenile salmonids are furunculosis, columnaris, coldwater disease (*Flexibacter psychrophilis*), pseudomonas/aeromonas, ichthyophthirius, nanophytes, and ceratomyxosis (*Ceratomyxa shasta*) (William Cox pers. comm.). Although infectious hematopoietic necrosis virus (IHNV) can cause losses of up to 100 percent of juvenile chinook and sockeye salmon in hatcheries and has been found to be symptomatic in almost all Pacific salmon, coho salmon do not appear to be susceptible (Wolf 1988).

The introduction of disease into wild stocks is becoming an increasing concern. The transmission of diseases from hatchery to native coho salmon stocks is a potential threat, but the degree of risk and seriousness of such a problem are little known (Brown et al. 1994). Although wild coho salmon may be exposed to, and become infected by, numerous parasites and microbial pathogens, BKD (caused by the bacterium *Renibacterium salmoninarum*) is the pathogen of greatest concern in California (Anderson 1995). *R. salmoninarum* is an obligate intracellular bacterial pathogen of salmonid fishes. Species particularly susceptible to the disease include brook trout and Pacific salmon, and especially coho salmon (Austin and Austin 1987; Inglis et al. 1993). The bacterium is slow growing, usually producing chronic disease in fish older than six months of age (Post 1987). Bacteria replicate and survive within host defense cells (macrophages) and the yolk of eggs, and are thus protected from the fish's immune defenses (William Cox. pers. comm.). *R. salmoninarum* survives for short periods in freshwater and seawater (approximately 14 days) (Austin and Austin 1987). BKD is widespread throughout the Pacific Northwest, and in California it has been detected at one time or another at most salmonid hatcheries and in wild salmonid stocks (Austin and Austin 1987; Inglis et al. 1993; William Cox pers. comm.).

R. salmoninarum is transmitted vertically (mother to egg) and horizontally (in water from fish to fish, or orally in hatcheries via unpasteurized fish feeds) (Warren 1991). Eggs are often infected from females that have high concentrations of *R. salmoninarum* in their ovarian fluids at spawning (Warren 1991; William Cox pers. comm.). The bacteria may infect eggs by passage through the micropyle. Probably less common, eggs may become infected while in the ovarian tissue, before exposure to ovarian fluid. Although males are infected by *R. salmoninarum*, they do not appear to play a role in vertical transmission (William Cox pers. comm.).

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There are essentially no treatments for BKD in wild populations of coho salmon due to consumptive and water quality issues (William Cox pers. comm.). Hatchery stocks have been successfully treated, however, once BKD is detectable in the fish it is impossible to eliminate or cure the disease. At best certain antibiotics can prevent progression of disease and transmission to uninfected fish. Once therapy is discontinued the disease usually resumes. Erythromycin has been used with good results, since it deposits in fatty tissues (e.g. egg yolks) and has a long half life in tissues. Adults injected approximately one month pre-spawning have produced clean eggs at Big Creek Hatchery (William Cox pers. comm.), and in various other hatcheries in the Pacific northwest.

Predation

Freshwater Predation

Anadromous salmonids have historically coexisted with both marine and freshwater predators. Predation occurs on all life stages of coho salmon, and though predation does not appear to have a major impact on a healthy population, it can be detrimental on those with low numbers or poor habitat conditions (Anderson 1995). As the quality of riverine and estuarine habitat decreases, predation may increase, playing a larger role in reducing some salmonid stocks as the loss of refuge habitat (e.g., deep pools, estuaries, LWD, and undercut banks) increases. Low stream flows and warmer water temperatures due to water diversions, water development, and habitat modification can enhance predation opportunities. These conditions may effect salmon mortality directly through predation, or indirectly through stress and disease, making them more vulnerable. Reduced water flow through reservoirs has increased juvenile travel time and thereby increased their exposure to predators (Columbia Basin Fish and Wildlife Authority 1991, as cited by NMFS 1998).

Predators such as invertebrates, fish, and birds, depending on conditions, can reduce the survival of eggs and alevins (Sandercock 1991). The vulnerability of this life stage to these animals depends on their depth and placement in the redd. If they are washed free of the redd, usually both stages are consumed quickly.

Freshwater fish predators of coho salmon are both native and non-native. Some of the native fishes known to consume coho salmon are: sculpin (*Cottus sp.*), steelhead rainbow trout (*Oncorhynchus mykiss*), coastal cutthroat trout (*O. clarki clarki*), and coho salmon (Shapovalov and Taft 1954; Sandercock 1991; Anderson 1995). Non-native or introduced fishes such as Sacramento pikeminnow (*Ptychocheilus grandis*), smallmouth bass (*Micropterus dolomieu*), and channel catfish (*Ictalurus punctatus*) can consume significant numbers of juvenile salmon if the conditions are favorable for them (NMFS 1998).

For example, Sacramento pikeminnow, a species native to the Sacramento and Russian river basins (Moyle 1976), were illegally introduced to the Eel River basin via Lake Pillsbury during the early 1980s (NMFS 1998). In just over ten years, they have spread to most areas of the Eel River basin, reflecting the fact that this ecosystem has been so significantly altered that this species now appears to be better adapted than native salmonids due to the artificially warmer water conditions (Brown et al. 1994). As a result, introduced Sacramento pikeminnow constitute a serious problem for Eel River system native salmonid populations (Higgins et al. 1992; CDFG

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1994a). If increased water temperatures and altered ecosystem trends continue, a shift towards the dominance of warmwater species can logically be expected (Reeves 1985).

Striped bass (*Morone saxatilis*) can also be a significant predator of juvenile salmonids, and have been observed in the Russian River system. However, there is no indication that they have had a significant impact on coho salmon.

Avian predators of juvenile salmonids include dippers (*Cinclus mexicanis*), gulls (*Larus spp.*), belted kingfishers (*Megaceryle alcyon*), herons (*Ardea spp.*), common mergansers (*Mergus merganser*), and osprey (*Pandion haliaetus*) (Sandercock 1991; Spence et al. 1996). Wood (1987) estimated that common mergansers were able to consume 10% of the coho salmon production in two coastal British Columbia streams. He also indicated the birds select fish by size, concentrating on girth rather than length, and feed primarily on hatchery fish, leaving the smaller native fish. As the quality of riverine and estuarine habitat decreases, avian predation will likely increase. Among mammalian predators that can impact salmonid populations, mink (*Mustela vison*) and otter (*Lutra canadensis*) can take significant numbers of the overwintering coho salmon juveniles and migrating smolts, although this is dependent upon conditions favorable to predators and the availability of other prey (Sandercock 1991).

Botkin et al. (1995) stressed that predation rates overall should be considered a minor factor in salmonid decline; yet as habitats are altered for both predator and prey, this may produce certain localized circumstances where predation has a greater impact.

Marine Predation

The relative impacts of marine predation on anadromous salmonids is not well understood, though documentation of predation from certain species is available. NMFS (1998) noted that several studies have indicated that piscivorous predators may control salmonid abundance and survival. Beamish et al. (1992) documented predation of hatchery-reared chinook and coho salmon by spiny dogfish (*Squalus acanthias*). Pacific hake (*Merluccius productus*) and pollock (*Theragra chalcogramma*) are known to consume salmon smolts (Holtby et al. 1990). Though not a major part of their diet, marine sculpins also consume juvenile salmonids.

There are many known avian predators of juvenile salmonids in the estuarine and marine environments. Some of these include belted kingfisher, gulls, grebes and loons (*Gavia spp.*), ardeids (herons, egrets, bitterns) cormorants (*Phalacrocorax spp.*), terns (*Sterna spp.*), mergansers (*Mergus spp.*), pelicans (*Pelecanus spp.*), alcids (auklets, murrelets, murrelets, guillemots, and puffins), sooty shearwaters (*Puffinus grisens*) (Emmett and Schiewe 1997; NMFS 1998). Bald eagles (*Haliaeetus leucocephalus*) and osprey are predators of adult salmonids (Emmett and Schiewe 1997). It is important to note that these animals are opportunistic feeders, meaning they will prey upon the most abundant and easiest to catch.

In the marine environment, the increase in marine mammal numbers, especially harbor seals (*Phoca vitulina*) and California sea lions (*Zalophus californianus*), has resulted in more encounters with the commercial and sport fishery and their gear (NMFS 1988). The effect of these interactions has at times been adverse to fishing harvest and equipment.

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According to Bokin et al. (1995), investigators in the early 1900s believed that seals and sea lions had a minor impact on the salmon declines. Stomach contents of hunter-killed animals that were believed to be killing salmon, rarely contained any salmon. Further, they analyzed numerous, more recent pinniped-salmon studies, reporting their strengths and weakness, and dismissed many as not statistically valid. The studies that were not dismissed indicated that marine mammal predation on anadromous salmonid stocks in southern Oregon and northern California played only a very minor role in their decline.

Hanson (1993) reported that foraging of California sea lions and harbor seals on anadromous salmonids was minimal at the mouth of the Russian River. Roffe and Mate (1984) found that pinnipeds fed opportunistically on fast-swimming salmonids, yet found they took less than one percent of the returning adult summer steelhead in the Rogue River, Oregon. Williamson and Hillemeier (2001a, 2001b) indicated that in the Klamath River estuary, pinniped predation rate estimates on coho salmon in 1998 and 1999 were 0.2 % and 1.2 % respectively. Hanson (1993) stated that predation on salmonids appeared to be coincidental with their migrations rather than dependent upon it.

Hart (1987) and Stanley and Shaffer (1995) studied harbor seal and salmonid interactions in the Klamath River estuary to evaluate the feeding activity of the seals during the Department's chinook, coho, and steelhead seining and tagging operations. They observed that salmonid predation increased on days when seining occurred, with little to no predation on non-seining days. The estimated percentage of seined and tagged fish taken by seals was relatively constant, ranging from about 3% to 8%. Hart (1987) observed that a majority of the fish were consumed by as few as 12 seals. This study exemplified pinniped opportunistic feeding habits: the seals consumed salmon that were likely made more vulnerable to predation through seining, handling, and tagging.

In most cases, salmonids appear to be a minor component of the diet of marine mammals (Scheffer and Sperry 1931; Jameson and Kenyon 1977; Graybill 1981; Brown and Mate 1983; Roffe and Mate 1984; Hanson 1993; Botkin et al. 1995; Goley and Gemmer 2000; Williamson and Hillemeier 2001a, 2001b). An analysis of scat samples of pinnipeds in the Smith, Mad, and Eel rivers found that salmonids had a relative abundance of 0.2% to 1.6% in the seals' scat (Goley and Gemmer 2000). Principal food sources of marine mammals include lampreys (Jameson and Kenyon 1977; Roffe and Mate 1984; Hanson 1993), benthic and epibenthic species (Brown and Mate 1983; Hanson 1993), and flatfish (Scheffer and Sperry 1931; Graybill 1981; Hanson 1993; Goley and Gemmer 2000; Williamson and Hillemeier 2001a, 2001b)).

Although salmonids appear to make up a relatively minor component of the diet of pinnipeds, this does not indicate conclusively that pinniped predation is not significant. The fact that coho salmon make up a small portion of an animal's diet could have no relation to the impact of that predation on the prey population. In fact, when a prey population is low, a very small amount of predation pressure can have a significant impact on the population. Predation may significantly influence salmonid abundance in populations when other prey are absent and physical habitat conditions lead to the concentration of adult and juvenile salmonids in small areas (Cooper and Johnson 1992). However, Cooper and Johnson (1992) also noted that based on catch data, some of the best catches of coho, chinook, and steelhead along the U.S. Pacific Coast occurred after marine mammals, kingfishers, and cormorants were fully protected by law.

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Native predators are part of the natural environment in which coho salmon evolved. However, the combination of increased predator populations and large scale modifications of habitat that favor predators can shift the entire predator-prey balance. Adult salmonid injuries resulting from marine mammal attacks were thought to be on the order of a few percent annually prior to 1990 (NMFS 1998). Predation may have an impact on abundance of salmonid populations where altered ecological conditions (e.g. increase in water temperatures) favor an introduced predator or physical constraints (e.g. restricted entrance to a fish ladder) lead to the concentration of adults or juveniles in small areas.

Hatcheries

Hatchery- and natural-origin coho salmon are of the same species. Coho salmon runs that are influenced by a hatchery may contain any combination of natural-origin, hatchery-origin, and naturalized hatchery fish. Even if hatchery- and natural-origin coho salmon are different in some ways, hatchery-origin fish represent an important component of the total species' gene pool. Hatchery-origin and natural coho salmon are often indistinguishable genetically. Hatchery stocks can be important for recovery.

Hatcheries are inherently neither good nor bad. Hatcheries have for many years provided significant societal and economic benefits. Many of the effects that are discussed in detail in this section can be negative. These include changes that occur in fish taken into the hatchery, effects of hatchery fish on natural stocks, and complications to monitoring natural populations. However, hatcheries can also be beneficial in a number of ways, including:

- C Conservation hatchery programs such as those at Warm Springs and Big Creek hatcheries have the potential to assist recovery of severely depleted coho salmon stocks in the Russian River and in streams south of San Francisco;
- C Supplementation of natural stocks by hatcheries may reduce extinction risk over the short-term by a) buffering the effects of small population size against environmental variation and catastrophic cohort failure, and b) by potentially accelerating recovery;
- C Carefully designed hatchery programs may increase the effective population size of a small population of natural spawners (e.g. Hedrick et al. 1995; 2000); and
- C Hatcheries aid in successful recovery of natural stocks by providing fish for controlled research (e.g. on disease) that could not be done using naturally-produced fish.

The specific impacts of hatchery-origin fish and hatchery practices and management on California coho salmon have not been well studied. However, many studies from inside and outside of California have addressed the differences between hatchery and wild anadromous salmonids and potential hatchery impacts. Based on this documentation, classes of effects can be reasonably evaluated for California coho salmon. The majority of the information described below is from studies outside California. The Department believes that the classes of effects described here may apply to California coho salmon to the extent that conditions and practices described in these studies are similar to those in California. For example, some of the studies described below were conducted at enhancement facilities in Washington and Oregon, that release large numbers of juveniles. The effects of these large scale releases may be different

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than would be observed due to the limited releases from smaller mitigation facilities found in California. Where possible, we have attempted to qualify the available studies as to their applicability to California coho salmon hatcheries using this criterion.

The following review considers the possible effects of hatcheries on California coho salmon. The Department has in recent years made significant changes to its coho salmon hatchery programs in order to incorporate considerations for conservation. These measures include:

- Non-native source stocks and interbasin transfers were never as extensive in California as in other Pacific Coast states with coho salmon hatcheries. The department has since the 1980s stopped all interbasin and out-of-state transfers of coho salmon.
- Hatchery production of coho salmon in California is a small proportion of the total Pacific Coast hatchery production of coho salmon. Coho salmon production has been reduced or eliminated at most of the recently active hatcheries in California.
- The department participates in two state-of-the-art conservation hatchery programs for coho salmon that are an important part of the recovery planning for stocks in the Russian River and streams south of San Francisco. A third conservation hatchery program for coho salmon is planned at Mad River Hatchery.
- The Department is in the process of producing Hatchery and Genetic Management Plans (HGMP - see Chapter VII *Hatchery Operations*) for all of its hatcheries that will incorporate a conservation mandate into hatchery operations and practices.
- The Department continues to develop hatchery goals and constraints for coho salmon production at its facilities and modernize hatchery practices to minimize hatchery effects on both the hatchery and natural stocks. All coho salmon released from California facilities have been marked since 1996.

Legislative direction in FGC sections 6901 and 6902 and Commission policies on salmon state that natural production are the foundation of the state's salmon resources. The following discussion is presented for the purpose of examining the relationship between natural stocks and the hatchery stocks that may contribute to and interact with them, and the effects of hatcheries and their operation on both hatchery and natural fish. The long-term viability of both hatchery and natural stocks may be affected by the interactions between them. Lessons learned through review of this information will guide future hatchery activities to increase the recovery rate of depressed coho salmon stocks.

Overview of the Effects of Artificial Propagation and Non-native Source Stocks

There is a large body of theoretical and empirical evidence that supports the conclusion that artificial propagation itself can and sometimes does negatively affect natural and hatchery salmonid populations (Steward and Bjornn 1990; Hindar et al. 1991; Waples 1991b; Campton 1995; Flagg et al. 2000). Several published studies have found that hatchery stocks are generally less productive in the wild than locally adapted natural stocks, and that transplanted stocks are

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less productive than locally adapted natural ones (Leider et al. 1990; Waples 1991b; Meffe 1992; Fleming and Gross 1993; Reisenbichler and Rubin 1999).

In many cases, hatchery-origin salmonids differ in significant and often heritable ways from wild fish (Table 9; Flagg et al. 2000). Hatchery environments are very different from the natural environments of streams and rivers. Hatchery protocols often do not adequately mimic spawning, growth profiles, survival profiles, social and learning environment, or emigration timing of natural runs that can negatively affect post-release performance and subsequent fitness and productivity of both natural and hatchery stocks. The literature generally shows that hatchery-origin fish exhibit greater egg-to-smolt survival, poorer post-release survival, impaired foraging ability, rearing-dependent and genetically-influenced differences in agonistic behavior, differences in habitat preference, increased risk-taking behavior and associated increased vulnerability to predation, reduced fright response, alteration of stream-adaptive cryptic coloration, non-adaptive morphology and physiology, poorly developed secondary sexual characteristics and impaired mate competition, and altered spawning time. These differences can have potential negative consequences on the fitness and productivity of both hatchery and natural stocks (Weitkamp et al. 1995; Flagg et al. 2000; NMFS 2001a). Even if hatchery fish have reduced fitness in comparison with natural-origin wild fish, they can still negatively impact wild populations by their disruption of optimal natural x natural matings.

Non-native hatchery fish can come in contact with natural stocks in two ways. Imported stocks can be released onto natural stock and spawn with them, and/or imported stock can stray from the release stream to other streams and interbreed with natural stock. In addition, either natural-origin or hatchery-origin stocks may stray to another hatchery where they become incorporated into a hatchery stock that may subsequently commingle with a natural one. Based on what is known about stock transfers, sources, and outplanting, all of these could have potentially happened to California coho salmon.

Hatchery straying has recently been of concern to salmon biologists and managers (Grant 1997; CDFG/NMFS 2001). Whereas homing enables local adaptation and provides opportunity for divergence among spawning populations, migration between populations encourages convergence. Straying can be defined as naturally- and hatchery-produced fish spawning somewhere other than their natal area (CDFG/NMFS 2001). The term is also commonly used to describe hatchery-stocked fish returning to a spawning site in a stream other than the one in which they were planted. Sometimes hatchery-origin fish that spawn naturally instead of returning to the hatchery may also be called “strays”. The proportion of a population that strays varies considerably among coho salmon populations, even over a small geographical area (Quinn 1997). However, even a small proportion of straying in a large hatchery stock can strongly affect the composition of salmon populations in receiving watersheds. The contribution of hatchery stock to the spawner population determines the level of impact, not the proportion of hatchery fish that stray (Nicholas and Van Dyke 1982; Grant 1997). This can affect not only the genetic composition of nearby stocks but can also severely compromise accurate stock assessment (see below, *Overharvest and Masking of Declines in Abundance*).

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Table 9. Summary of differences between wild (W) and hatchery (H) salmonids (based on review by Flagg et al. 2000 with modification and additions).

General category	Specific category	Difference	Literature examples
Survival	Egg-to-smolt survival	H > W	Leitritz and Lewis 1980; Piper et al. 1982; Pennell and Barton 1996
	Post-release survival	H < W	Greene 1952; Salo and Bayliff 1958; Miller 1953; Mason et al. 1967; Fraser 1981; Fraser 1989; LaChance and Magnan 1990 a, 1990b; Reisenbichler and McIntyre 1977; Chilcote et al. 1986; Leider et al. 1990
Foraging behavior	Ability to effectively forage in natural environments	H < W	Sosiak et al. 1979; O'Grady 1983; Myers 1980; Mason et al. 1967; Uchida et al. 1989; Johnson and Ugedal 1986
Social Behavior	Agonistic behavior	Outcome depends on rearing environment	Symons 1968; Bachman 1984; Uchida et al. 1989; Grant and Kramer 1990; Olla et al. 1990; Berejikian 1995 a, 1995 b; Olla et al. 1998; Moyle 1969; Swain and Riddell 1990 ^h
Habitat preference	Location in water column and orientation to the water surface	H higher than W H more surface oriented than W	Dickson and MacCrimmon 1982; Sosiak 1978; Mason et al. 1967; Uchida et al. 1989
Response/exposure to predators	Level of risk-taking behavior Fright response	H > W H < W	Uchida et al. 1989; Maynard et al. 1995; Olla et al. 1998; Johnsson and Abrahams 1991; Berejikian 1995b; Mason et al. 1967
	Cryptic coloration	H < W	Donnelly and Whoriskey 1991, 1993; Maynard et al. 1996
Morphology and physiology	Morphology	H morphology different from W, H swimming speed < W swimming speed	Taylor and Larkin 1986 ^h ; Bams 1967; Taylor and McPhail 1985 ^h
	Physiology	Stress in presence of predators during smolting (H > W)	Järvi 1990
Reproductive behavior	Secondary sexual characteristics and ability to compete for mates Primary sexual characteristics	H < W H > W	Fleming and Gross 1989 ^h , 1992 ^h ; Berejikian et al. 1997 ^h
	Change in spawning time	H spawning time often earlier than W	Flagg et al. 2000; Nickelson et al. 1986 ^h

^h Results specific to coho salmon

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Although it is frequently assumed, it is unclear based on the available information whether hatchery salmon stray at a greater rate than natural-origin salmon; in some studies (all from outside California) they did and in others they did not (McIsaac 1990; Jonsson et al. 1991; LaBelle 1992; Potter and Russell 1994). Also, studies to date may have been too limited to draw accurate general conclusions from them (Waples 1999). Outplanted salmon might stray more than locally reared and released salmon, and straying fish may tend to enter nearby rivers more than distant ones, although there are many exceptions (reviewed in Quinn 1993; Grant 1997; CDFG/NMFS 2001). Quinn (1997) summarized the issue by saying that salmon tend to return to their release site, then on to their rearing site, if that site can be detected. If it cannot be detected, then they tend to spawn in the nearest appropriate place.

In a study specific to coho salmon, LaBelle (1992) did not find a significant difference in stray rates in coho salmon hatchery and wild populations on Vancouver Island. This author also observed age specific stray rates in coho salmon (older fish stray more), and suggested that coho salmon may home better to their natal site than to a new site. In a comparative study of straying, Shapovalov and Taft (1954) found that coho salmon strayed more than steelhead in two streams south of San Francisco. Stray rates for natural coho salmon in that study (15-27%) were considerably higher than for several Vancouver Island natural coho salmon populations (LaBelle 1992; 0-3.9%). Estimated stray rates of Trinity River Hatchery-produced coho salmon averaged 54.5% between 1997-1999 (reviewed in CDFG/NMFS 2001). The annual straying estimates were variable: 75.8% in 1997, 57.0% in 1998, and 30.8% in 1999. In several studies (Vreeland et al. 1975 [Washington], Solazzi et al. 1991 [Oregon]), coho salmon that were trucked from a hatchery to a release location tended to have impaired homing to the hatchery. Stray rates increased with distance between the rearing and release sites in Solazzi et al. (1991 [Oregon]). Stray-rate estimates from different studies are not directly comparable. However, to illustrate the kind of variation seen in these studies, some published stray rates for coho salmon are listed in Table 10.

Table 10. Some stray-rate estimates for hatchery and natural coho salmon populations.

Population	Range of Stray Rates Observed (Percent)	Literature Source
California, natural	15-27	Shapovalov and Taft 1954
Puget Sound, natural	1-65	Vander Haegen and Doty 1995
Washington coast, natural	0-67	Vander Haegen and Doty 1995
Columbia River, hatchery	0-12.4	Vander Haegen and Doty 1995
Washington coast, hatchery	< 0.5-4	Vander Haegen and Doty 1995
British Columbia, hatchery	0-27.7	LaBelle 1992

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Overharvest and Masking of Declines in Abundance

In the presence of a large hatchery-origin component, natural-origin stocks can experience harvest pressure beyond that which they can support. High exploitation rates that minimally affect the hatchery stock component can overharvest the natural one (Ricker 1981; McIntyre and Reisenbichler 1986; Lichatowich and McIntyre 1987). Nehlsen et al. (1991) identified overharvest of natural stocks in mixed fisheries as a widespread factor in the decline of natural stocks; 100 of the 214 natural spawning stocks identified as being at risk of extinction were affected by overharvest in mixed fisheries. Coho salmon are not currently subject to an ocean harvest season in California waters, although they are included in by-catch in the chinook salmon ocean fishery. However, overharvest of natural-origin coho salmon in the mixed ocean salmon fishery may have been a significant factor in the history of the decline of California coho salmon prior to the fishery closure in 1993 (Brown et al. 1994).

Abundant hatchery-origin stocks can also mask the decline of natural-origin stocks. Historically, many, if not most, hatchery-origin coho salmon from California facilities were not marked before release. When these fish returned, they were superficially indistinguishable from natural-origin fish. Because of this, the true proportion of hatchery returns relative to natural-origin returns is not known for certain. Hatchery influenced runs may be composed of natural-origin, hatchery-origin, and naturalized hatchery-origin fish or a mixture. The composition of these mixtures is usually unknown (but see Nicholas and Van Dyke 1982, and discussion above).

The Department estimated that natural-origin coho salmon comprised 66% of their total Klamath River estuary juvenile coho salmon catch in 1997, 39% in 2000, and 27% in 2001 (CDFG 2000, 2001d). In addition, hatchery-origin coho salmon were larger (mean FL 150-160 mm) compared to natural origin coho salmon (mean FL 120-130 mm).

Hatchery supplementation¹³ programs are designed to contribute spawners to the natural population, but have generally been unsuccessful. In the vast majority of examples (as reviewed in Flagg et al. 2000; Miller et al. 1990), salmonid supplementation programs have failed to produce a self-sustaining run. California coho salmon facilities are all either mitigation facilities (compensating for lost spawning habitat) or fishery enhancement, not supplementation. However, they likely do contribute some hatchery-origin spawners to the streams where they are located and those nearby. Brown et al. (1994) noted decreased abundance of coho salmon with increasing distance from production facilities. This suggests that hatchery-origin fish may inflate estimates of coho salmon abundance when hatchery and natural-origin escapement is combined, the effect being greatest near release sites and decreasing with increasing distance from the hatchery. At the same time, supplementation can theoretically enhance, and at times has demonstrated, an ability to support and contribute to small natural runs.

Modified supplementation programs based on the Conservation Hatchery Concept (Flagg and Nash 1999; Flagg et al. 2000) can overcome or minimize some of the problems associated with hatchery operations and hatchery fish to provide short-term, last-chance relief for

¹³ Supplementation hatcheries are intended to contribute to the natural spawning population. Mitigation hatcheries are intended to make up for reductions in natural spawning due to human-caused habitat loss (e.g. dam construction). Enhancement hatcheries are intended to improve a fishery by increasing the number of catchable fish in the ocean or stream. Conservation hatcheries are experimental programs intended to supplement depressed natural populations or provide fish for artificial recolonization of streams that have experienced extinctions. Conservation hatcheries attempt to minimize or eliminate negative effects common to fish culture, resulting in as close to wild fish as possible.

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populations on the brink of extinction. Currently, Warm Springs and Big Creek hatcheries in California are developing coho salmon conservation hatcheries based on captive broodstock.

With the exception of Trinity River and Iron Gate hatcheries, current coho salmon hatchery programs in California are either small, production has been terminated, or they are in the process of conversion to conservation hatcheries. However, NMFS (2001a) stated that past hatchery releases probably masked the true degree of decline of coho salmon populations in the CCC Coho ESU. NMFS (2001a) also stated that existing populations in CCC Coho ESU streams might only exist as a result of hatchery input.

Ecological Effects of Hatchery-Origin Fish on Natural-Origin Fish

Competition for food or space can occur when niches overlap, resources are limiting, and individuals are co-occurring in time and space. Intraspecific competition is generally thought to be more intense than interspecific competition because niches overlap more completely. Hatchery stocks, if released in large numbers relative to natural-origin juveniles in a limiting environment, may negatively affect natural-origin fish through competition. Many studies have found that hatchery-origin fish perform poorly after release (e.g., Miller 1953; Bachman 1984; Maynard et al. 1996; Flagg et al. 2000). However, other studies have suggested that hatchery-origin fish are competitively superior (e.g. when they are released at a larger size than the natural fish) and can displace natural-origin fish (Nickelson et al. 1986). Nickelson et al. (1986) reported that pre-smolt releases of Oregon hatchery-origin coho salmon were associated with displacement of natural coho salmon from their usual territories. Fraser (1969) reported depressed growth rates and increased mortality in coho salmon due to intraspecific competition for resources. Shapovalov and Taft (1954) found that for streams south of San Francisco the number of coho salmon outmigrants was inversely related to adult returns, suggesting that intraspecific competition somehow improves ocean survival of migrants. Emlen et al. (1990) and Ogura et al. (1989) discussed evidence for density-dependent factors affecting ocean survival in coho salmon. To the extent that these density-dependent factors hold in the ocean, increases in hatchery-origin coho salmon abundance have the potential to reduce the ocean survival of natural coho salmon. Competition can also occur among adults on the spawning grounds for space and mates.

Competition between Iron Gate Hatchery chinook salmon and natural coho salmon juveniles (as well as natural chinook and steelhead) due to early summer chinook releases was discussed in a recent review of California hatcheries (CDFG/NMFS 2001). Reduced river flows at this time of year and crowding of fish into the river from the hatchery increases the likelihood of competition among these stocks by concentrating fish at high densities within a few cold water refugia. The already high density of fish at these sites may be exacerbated by hatchery releases. Increased stress and disease transmission were other possible effects. Several alternate release strategies were proposed to alleviate these potential problems.

Other ecological factors that may impact coho salmon are cited in Flagg et al. (2000) and include predation by larger hatchery-origin fish on natural-origin fish, negative social interactions between hatchery and natural stocks, compromised fish health, and negative effects on migratory behavior. Waples (1991b) noted that wild fish are much more closely tied to climatic and environmental cues to outmigration than hatchery fish. Whereas large early flows may be sufficient to stimulate hatchery salmon to emigrate, wild or natural-origin salmon may respond

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better to prolonged or later flows that allow more variation in outmigration timing. Steward and Bjornn (1990) commented on the possibility of increased pikeminnow predation on coho salmon correlated with hatchery releases in the Columbia River.

A relatively common feature of hatchery stocks is alteration of run timing (Nicholson et al. 1986; Flagg et al. 1995). The normal cause is hatchery spawning of the first few fish that return rather than using broodstock collected over the entire run-time. This can change other ecological characteristics as well including body size and outmigration timing. Brown and Moyle (1991) noted that the Prairie Creek Hatchery coho salmon run returned earlier than the natural run. Precocious maturation, especially of males, is also a common negative feature of salmon propagation (Flagg et al. 2000).

Genetic Effects of Hatchery-Origin Fish on Natural-Origin Fish

Hatcheries are known to have the potential to affect the genetic integrity of natural populations when they come in contact with them (Simon et al. 1986; Withler 1988; Waples and Teel 1990). The genetic risks associated with hatcheries have been discussed extensively in the literature (Hindar et al. 1991; Waples 1991b, 1999; Busack and Currens 1995; Campton 1995; Allendorf and Waples 1996; NRC 1996). The fitness of natural-origin salmon can be decreased if they mate with hatchery-origin fish (Reisenbichler and McIntyre 1977; Reisenbichler 1997; Reisenbichler and Rubin 1999; Flagg et al. 2000). Campton (1995) noted that many of the effects attributed to hatchery fish are really due to hatchery and fishery management practice, rather than to the fish themselves. Over most of the history of hatchery influence on natural coho salmon stocks, genetic effects went unnoticed because they are subtle and hard to recognize unless genetic management is part of normal monitoring.

The effects of hatchery influence on wild population genetics are not always as severe or benign as expected since they depend largely on the differences between specific hatchery and wild stocks, and interbreeding or other interactions occurring between them. Hindar et al. (1991) and Skaala et al. (1990) in reviews of the genetic effects of hatchery stocks on wild salmonids cited examples of effects that ran the gamut from native stocks that had been largely or entirely displaced by hatchery stocks, to hybridization between native and hatchery fish, to examples in which repeated hatchery releases had no deleterious effect at all on the native population.

Waples (1991b) provides a framework for understanding the classes of impacts that hatchery fish/operations can have on the genetics of natural populations: 1) direct genetic effects due to hybridization of hatchery and natural fish and subsequent introgression; 2) indirect genetic effects due to altered selection regimes or decreases in population size caused by competition; predation, disease, or other factors not involving introgressive hybridization; and 3) genetic changes in hatchery stocks through artificial/natural selection in the hatchery, genetic drift, or use of non-native stock, which magnifies the consequences of hybridization when the stocks mix.

Introgressive hybridization between divergent hatchery and wild stocks is a direct genetic effect that can result in reduction of genetic variance between populations. In this case, the danger is that complete mixing of stocks will occur, resulting in a diversity of locally adapted wild stocks being replaced by a smaller number of relatively homogeneous ones (Allendorf and Leary 1988). Reisenbichler and Phelps (1989) found circumstantial evidence for the homogenizing effect of hatchery outplanting of steelhead in Washington.

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Another direct genetic effect, outbreeding depression, is defined as the mating of individuals from divergent populations that results in loss of fitness in subsequent generations (Lynch 1997). As the genetic distance between populations increases, the probability of genetic incompatibility increases. Outbreeding depression results when a locally adapted wild gene pool is swamped by genes from divergent hatchery fish. In this scenario, displacement with immigrant alleles from hatchery adapted or non-native stock cause disruption of adaptive gene complexes (Dobzhansky 1955). As the distribution of a homogeneous stock becomes more widespread, its negative effects become greater as it encounters distinctive wild stocks. As summarized in Waples (1991b), Emlen (1991) determined through modeling that, if selective regimes are sufficiently different, reductions in fitness can occur even if mixture proportions are small (5-10%), and that recovery of fitness lost due to a single hybridization event may require many generations. Outbreeding depression due to hatchery influence is documented as a factor leading to reduced productivity in some coho salmon populations. Nickelson et al. (1986) in an experiment with hatchery influenced and non-hatchery influenced coho salmon in Oregon, documented decreased natural production in supplemented streams. NMFS (2001a) stated that potential outbreeding of large numbers of hatchery coho salmon from Trinity River Hatchery poses a significant threat to the genetic integrity of natural populations in the watershed.

Outbreeding can increase heterozygosity¹⁴ in a stock. Mating with similar local stocks is sometimes suggested as a way to increase diversity within a reduced-diversity hatchery-origin stock (e.g., Simon et al. 1986). Bartley et al. (1992) reported a relatively high level of heterozygosity in hatchery influenced coho from Waddell Creek in comparison to the low heterozygosity reported for nearby Scott Creek, which had little hatchery influence.

Competition, predation, disease transmission, effects on ocean survival, and changing selective regimes can all affect genetic structure and cause changes in wild stocks indirectly through reduction of population size. Any factor that causes reduction in either total population size or effective population size (N_e)¹⁵ can affect genetic structure. Reviews of these factors can be found in Steward and Bjornn (1990), Flagg et al. (2000), and above. Reduction of total population size can result in increased risk of local extinction and changes in genetic structure due to altered demographic factors.

Genetic changes occur in hatchery stocks in four major ways (Campton 1995; Busack and Currens 1995; Waples 1999): 1) intentional or artificial selection for a desired trait (e.g. growth rate or adult body size); 2) selection due to non-random sampling of broodstock; 3) unintentional or natural selection that occurs in the hatchery environment; and 4) temporary relaxation of selection during the culture phase that otherwise would occur in the wild.

Reduction in effective population size due to small numbers of breeders (N_b) causes “erosion of genetic variability through random extinction of alleles” (Waples 1991b). Small effective population size increases the proportion of individuals that are homozygous for deleterious recessive traits. The resultant reduction in fitness, called inbreeding depression, is defined as “exposure of the individuals in a population to the effects of deleterious recessive genes through matings between close relatives” (Lynch 1997). Substantial inbreeding depression

¹⁴ In a population of organisms, the likelihood that an individual has different alleles for a given gene. A measure of genetic diversity.

¹⁵ Used in management of genetic resources to express information about expected rates of random genetic change due to inbreeding and genetic drift. The size of a hypothetical ideal population with the same amount of random genetic change as the actual population experiences. Typically the effective population size is lower than the total population size.

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has been reported in some hatchery stocks (reviewed by Allendorf and Ryman 1987). Utter et al. (1989) and Waples et al. (1990) did not find reduced heterozygosity in a survey of Pacific Northwest hatchery chinook. In California, Big Creek Hatchery used very small numbers of breeders (< 20 females and < 25 males per year) in their coho salmon hatchery program between 1994-1999 (see Chapter VII, *Hatchery Operations*).

Ryman and Laikre (1991) demonstrated that the overall effective population size of a mixed natural and hatchery population can be lower than that for the natural population alone. This decrease is especially likely if: 1) N_e of the natural population is small; 2) contribution from artificial production is large; and 3) N_e of the artificial population is small. With traditional hatchery operations and dwindling natural stocks, these conditions are often met. However, careful hatchery management (i.e. conservation hatchery management) can minimize reduction of overall N_e or even increase it (e.g. Hedrick et al. 1995, 2000).

Many of the differences between hatchery and wild salmonids (Table 9) that make hatchery fish different and less fit than wild fish in nature are due to different selective regimes in the hatchery, or substitution of selection leading to what is called domestication selection. Domestication selection can be defined as any change in the selection regime of a cultured population relative to that experienced by the natural population (Waples 1999).

Genetic change mediated by selection in hatchery populations is probably inevitable since selection will occur unless several unlikely coincidences occur that cancel differences in wild and hatchery mortality profiles (Waples 1991b, 1999). Some divergence between hatchery and natural stocks from which they were derived will always occur in hatcheries.

Natural selection that occurs in the hatchery includes selection for traits that are well adapted to hatchery conditions and avoidance of early life stage mortality that would normally occur in the wild. Early life-stage mortality may be as high as 50% in wild salmon, compared to 10% or less in hatchery salmon; but, post-release mortality of hatchery fish may be 99% or more, much higher than for wild fish (Howell et al. 1985). Artificial selection can also occur if broodstock are chosen to accentuate some trait that has perceived management or product value (e.g. age, size, time of return). Waples (1991b) states that, even if hatcheries attempt to control artificial selection, it will likely occur anyway since there is no way to mimic natural selection for reproductive success.

Conclusions

Brown et al. (1994) stated that most coho salmon stocks inhabiting large rivers in California are dominated by hatchery fish. Chapter VII, *Hatchery Operations*, reviews the available information on active coho salmon hatcheries in California, and this section reviews the possible and documented impacts associated with hatchery operations and hatchery fish. Based on this information, three conclusions can be drawn: 1) hatcheries have historically been active throughout the range of coho salmon in California; 2) California coho salmon hatcheries have produced numbers of fish that, while relatively small in a coastwide sense, are significantly large relative to natural production in places where large hatcheries have been active; and 3) stocks other than native ones, including out-of-basin and out-of-state imports, were propagated and released through California hatchery operations, and those returning fish clearly had the opportunity to interbreed with natural-origin coho salmon. These conclusions suggest that

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although hatcheries may have produced some benefits to local coho salmon populations, hatcheries have also had the opportunity to adversely affect natural California coho salmon populations.

Although the potential for negative hatchery impacts has existed in California for many years, and is implicated as a negative factor by available literature and documented production information, it is unclear exactly whether or how hatchery fish and/or hatchery operations have affected and are affecting California's natural coho salmon. First, the extent to which hatchery stocks interbred with natural stocks is not known. Second, the level of straying of hatchery stocks to streams with wild stocks is not known for all stocks¹⁶. Third, although many activities that have been shown to have negative impacts on salmonids in other places have occurred in California, specific negative impacts to coho salmon populations here are undocumented and therefore cannot unambiguously connect hatchery fish and/or hatchery practice to the decline of coho salmon in California.

Although no direct connection can be made due to lack of specific data, stock transfers from various sources from within and outside California have been implicated by several authors as a factor that might have contributed to the low diversity and weak population divergence observed in California coho salmon stocks (Brown and Moyle 1991; Bartley et al. 1992; Weitkamp et al. 1995; NMFS 2001a). Bartley et al. (1992) reported that hatchery influenced Waddell Creek coho salmon had the highest measure of heterozygosity of the 27 populations surveyed, possibly due to outbreeding with imported hatchery-origin stocks. They also suggested (and also discussed in Brown et al. 1994) that outbreeding with imported hatchery-origin stocks from within California, as well as from out-of-state sources, might have obscured whatever genetic differentiation formerly existed among California coho salmon populations. Waples (1991b) noted that historical mixing of stocks can often be detected as homogeneity of geographically distant populations. Bryant (1994) in a status review of coho salmon, noted that planting of non-native coho salmon in Scott and Waddell Creeks between 1950-70 likely contributed to the decline in returns and to the current genetic make-up of these stocks.

Hatcheries in California have dramatically reduced their production of coho salmon, limited outplanting, and stopped virtually all stock transfers in recent years. Therefore, current impacts of hatchery fish/operations on whatever remaining natural stocks still exist may be significantly less than in the past.

Widespread hatchery stocking should not be used by itself as documentation of wild stock extinction. Wild coho salmon stocks can persist in the presence of extensive hatchery stocking. Utter et al. (1995) reported the persistence of major ancestral regional patterns in Columbia River chinook salmon in the face of long non-native hatchery influence. Also, Phelps et al. (1994) unexpectedly found what appeared to be local native populations of rainbow trout in many places with long histories of non-native hatchery planting in Washington.

Many of the potential impacts reviewed here could have occurred given what is known about the universality of the results of research in this area. Hatcheries may have contributed to declines of coho salmon in California, although to what degree is unknown. Their potential to do harm is severely limited by decreased production and modern management policy.

¹⁶ However, the few available estimates of stray rates of natural (Shapovalov and Taft 1954) and hatchery (CDFG/NMFS 2001) stocks are relatively high.

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Genetic Diversity

Measures of Genetic Diversity

Genetic diversity within a species can be thought of in terms of the types and distribution of raw genetic material (i.e., alleles) that is present in individuals across the species' range. The structure of genetic diversity can be expressed at two levels: *within population diversity*—differences and similarities among individuals within a local breeding population, and *between population diversity*—differences and similarities among more or less separate breeding populations. The measurement and presentation of genetic structure has been discussed extensively in the literature leading to a number of more or less standard ways to interpret and present data (e.g. Nei 1987). *Within population* structure can be expressed using such measures as heterozygosity (see previous discussion) and number of alleles per locus. *Between population* genetic structure of wild populations is due mainly to the effects of reproductive isolation, genetic drift¹⁷, gene flow, and local adaptation through selection, acting separately and together, on the available genetic variation in spawning populations. The structure of genetic variation within and among salmon populations is hierarchical: at the base are substantially reproductively isolated local breeding populations, together these form metapopulations typically connected by some small amount of gene flow among the members, then larger biological races, then subspecies/ecotypes, and culminating with the species as a whole (NRC 1996).

Loss of genetic variation can mean loss of alleles, loss of heterozygosity, or changes in allele frequencies. All of these have the potential to be non-adaptive, and can negatively affect the character and persistence of breeding populations. The risks associated with loss of genetic diversity have been explored in a number of published papers including Waples (1991b), Currens and Busack (1995), Busack and Currens (1995), Campton (1995), Grant (1997), and Utter (1998). In addition, it is also important to draw a distinction between total genetic diversity and adaptive genetic diversity. The ability of a population to respond to change can be negatively affected by unique but maladaptive genes that nonetheless add to total genetic diversity.

The Importance of Genetic Diversity

Genetic resource conservation is as important to species preservation and persistence as is habitat conservation. Conservation biologists argue that biodiversity (and its genetic underpinnings) should be conserved for three reasons (McElhany et al. 2000; Levin and Shiewe 2001). Firstly, diversity leads to greater abundance because different populations can exploit different habitats and resources. The diversity of salmon life history (e.g. run timing over the range of chinook salmon) and its underlying genetic components are a good example of this. Secondly, Diversity fosters enhanced long-term stability by spreading risk and providing redundancy in the face of unpredictable catastrophes, e.g. due to climatic or ocean condition fluctuation. Environmental challenges to natural populations are often dramatic and sudden (e.g. El Nino events). Because of this, loss of diversity can depress the potential of the entire resource to respond to environmental change. These factors clearly apply to salmon. Finally, genetic diversity provides a range of raw material that allows adaptation and increased probability of persistence in the face of long-term environmental change.

¹⁷ Genetic drift is a random change in allele frequency that occurs in small populations.

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Loss of variation due to inbreeding depression has been reported as a factor that may increase the probability of local extinction (Saccheri et al. 1998). Loss of variation has also been implicated as a factor limiting evolutionary potential (Frankham et al. 1999), and can affect the potential range of response to pathogens (O'Brien and Everman 1989).

Factors That Reduce Genetic Diversity and Conservation Guidelines

Many of the causes of genetic diversity loss are related to decreases in population size and associated decreases in effective population size and number of breeders (N_b). Per generation loss of genetic diversity is related to the effective population size of the spawner population. N_e is usually much less than total population size (N_t); N_e/N_t estimates of 0.1-0.33 are thought to be reasonable for salmonids (Hedrick et al. 1995, based on original estimates by Bartley et al. 1992; Robin Waples, pers. comm.) and are often used for purposes of estimating population size targets for conservation (McElhaney et al. 2000). Several authors have proposed N_e thresholds that can be used as guidelines in evaluating the severity of potential genetic diversity reductions. Effective population size of 50 was proposed by Franklin (1980) as the lower limit to avoid inbreeding depression. Waples (1990) suggested that short-term maintenance of genetic variation in salmon could be achieved with 100 effective breeders per year based on the probability of losing rare alleles. A minimum N_e of 500 is thought to be enough to avoid long term loss of genetic variation (Franklin 1980; Lande and Barrowclough 1987). Lynch (1990) wrote that an effective size of about 1,000 is usually large enough to maintain genetic variation in a population. N_e of 5,000 may be sufficient to maintain potentially adaptive genetic variation (Lande 1995).

Because salmon populations are usually connected by some small amount of gene flow, and gene flow between populations is a contributor to overall genetic variation, smaller than predicted effective sizes might be sufficient to maintain diversity. Also, estimates from two of the studies above (Franklin 1980 and Lande 1995) were based on study of a single species, the fruit fly *Drosophila melanogaster*, and might not be generally applicable to salmon (McElhaney et al. 2000).

Using the estimate of reasonable N_e/N_t ratios above and the average generation length for the species, one can arrive at targets for effective population sizes per generation and annual spawner abundance sufficient to avoid loss of genetic variation. Applying the lower end of the range of reasonable N_e/N_t ratios (0.1) to the range of minimum sizes from the literature cited above (500-5,000), the target minimum *population size per generation* sufficient to maintain long-term genetic variation ranges from 5,000 (Franklin 1980) to 50,000 (Lande 1995). Coho salmon in California almost all spawn at age 3 giving an average generation length of 3 years. Therefore, a rough estimate of the minimum number of coho salmon *breeders per year* necessary to maintain genetic diversity and ensure long-term persistence is 1,667 to 16,667. Clearly, many local breeding populations of coho salmon in California do not fall within this range. Therefore, the potential for loss of genetic variation in California coho salmon appears to be high.

Another factor that can reduce genetic diversity and fitness is introgressive hybridization¹⁸ of different stocks due to straying and artificially high levels of gene flow which may cause locally adapted populations to be more similar to one another and less well adapted to the place

¹⁸ Hybridization in which offspring of hybrid individuals mate with some level of success causing population mixing of extraneous genes with local ones.

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where they live. Hybridization can also affect productivity. Much of the discussion in the literature regarding loss of diversity has been in the context of impacts associated with hatchery management and practice, and interactions of hatchery with natural fish. The extent to which introgressive hybridization has affected California coho salmon stocks is unknown.

Declines in abundance have undoubtedly resulted in losses of genetic diversity in salmon. For example, winter-run chinook salmon in the Sacramento River have the lowest genetic diversity (i.e. fewer alleles at most loci and lower average heterozygosity) of the four runs that exist there (Banks et al. 2000) likely due to severe historical reductions in abundance. Coho salmon have been reported to have the lowest genetic diversity of the five Pacific salmon species (Allendorf and Utter 1979; Waples et al. 2001). Some studies of California coho salmon have found evidence of low genetic diversity (Olin 1984; Bartley et al. 1992). One explanation of this may be that coho salmon have undergone one or more severe reductions in population size.

Habitat Condition

Forestry Activities

Introduction: Forestry practices have been shown to impact several freshwater habitat components important to anadromous salmonids in general, and coho salmon specifically. These impacts include: increased maximum and average summer water temperatures, decreased winter water temperature, and increased daily temperature fluctuations; increased sedimentation by fine and coarse sediments; loss of LWD; decreased DO concentrations; increased in-stream organic matter; and decreased stream bank stability (Salo and Cundy 1987; Meehan 1991; Moring et al. 1994; Murphy 1995; Monschke 1996). Even when some habitat conditions return to pre-timber-harvest levels, fish populations do not always recover, which may be due to other habitat conditions remaining sub-standard or having been permanently altered (Moring et al. 1994). Harvest-altered areas are further affected and aggravated by natural (e.g. blow downs, naturally-caused landslides) and other human-related activities, thus resulting in cumulative effects to coho salmon and their habitat.

Existing information combined with that from other states and that on effects to anadromous salmonids in general, reveals that forestry practices can have deleterious effects on coho salmon. Data from northern California corroborates research from other regions in the Pacific northwest. Nakamoto (1998) found that juvenile coho salmon density dropped the most during and after logging, did not recover after three years, and biomass was less after, rather than during, harvest operations. Krammes and Burns (1973) found that smolt biomass decreased, though fry biomass increased, after road construction. Clearly, there are effects that are legacy impacts (see below). These impacts include increased in-stream sediment load, upslope erosion, loss of LWD in streams and reduction of future upslope supply, and removal of stream-side vegetation.

The effects of forest activities on coho salmon are complex. Alterations to habitat and direct effects to this species arise from many factors, including the long history of logging in the coastal watersheds, the different activities affecting the habitat components, individual activities affecting multiple habitat components, the interrelatedness of the habitat components themselves, and the changes in timber harvest over the many years of logging. Forestry practices have been linked to important changes in watersheds and stream habitats that affect anadromous salmonids

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(Meehan 1991; Murphy 1995). These changes include increased sedimentation and water turbidity, increased water temperature, loss of stream habitat complexity, loss of in-stream woody debris and upslope debris supply, and altered stream flow and water supply.

Identifying the relationships between forestry practices and habitat impacts is complicated for several reasons. First, there is a long history of timber harvesting, and some effects, such as sedimentation and slope instability, continue long after harvesting has occurred. These alterations are referred to as “legacy” effects, and recovery may take many decades (Murphy 1995). Legacy effects are a factor along the north coast of California (Monschke 1996). Second, there have been many technological and management changes in timber harvest, and it is difficult to differentiate legacy effects from recent or current effects. Third, the salmonid habitat elements affected by timber harvest are themselves intimately inter-related. The amount and distribution of LWD, water temperature, near-stream vegetation, sediment transport and deposition, landsliding, stream flow and supply, and turbidity are all linked to one another. Hicks et al. (1991), Seddell et al. (1991), Ligon et al. (1999) state that the lack of quantitative data, long-term studies, the varying temporal and spatial scales of harvest practices, and the sparse number of site-specific studies result in difficulty in interpreting the effects on anadromous salmonids. Nonetheless, these authors acknowledge that forest practices impact both anadromous fish habitat and fish populations directly.

Monschke (1996) found that different harvesting practices resulted in very different effects on salmonid habitat. He found that activity in the riparian zone, supply and delivery of LWD, sedimentation, and sediment transport were inter-related. He also found that recovery of canopy vegetation, upslope, and in-stream LWD components, and recovery from sedimentation and erosion effects took place at different rates. Canopy re-closure and stream-side revegetation of the riparian corridor was relatively quick. Conditions for stream sediment were variable. The natural removal of built-up, excess sediment, and effective reduction in sediment input from harvest activities was slow but measurable. However, sediment elements continued to impact both spawning and rearing habitat for anadromous salmonids. Lastly, the ability of a watershed to supply high-quality LWD (i.e. conifers) in the short-term was eliminated and long-term supply was thought to require considerable time. Thus, the total recovery of habitat components necessary for coho salmon was going to take considerable time.

The inter-relatedness of LWD, sediment storage and transport, water yield, and water quality complicate the interpretation of forestry practices and alterations in coho salmon. Lisle and Napolitano (1998) found that timber harvest resulted in sediment transport, increased delivery of LWD following blow-downs, and increased water yield. These factors resulted in a net increase of stored sediment and a greatly increased number and volume of pools, even without a net increase in bed transport. Complicating the observations was that the water yield was thought to be too modest to be the cause of sediment source or scouring of upstream pools. The key was thought to be the increased delivery of LWD. The LWD created more pools and trapped sediment from moving further downstream. The researchers went on to state that the current stream conditions beneficial to anadromous salmonids, high pool volume and frequency and stream diversity, were temporary. The stream conditions likely would be different under more extreme flows, and future LWD recruitment would be below normal. Hence, future pool habitat would be altered and degraded, affecting sediment accumulation, thermal refugia, and pool scouring. These findings reflect the complicated relationship of habitat components, the particulars of the timber harvest, and climate.

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Timber harvest has been occurring in the northwestern region of California since the mid-nineteenth century and continues in the watersheds both historically and currently inhabited by northern California coho salmon. During the approximately 150-year history of timber harvest in coastal northern California, harvest practices have changed dramatically, primarily due to changes in technology and decreasing availability of larger or higher quality logs. Where historical harvest and milling were close to waterways, modern trucks and tractors enabled harvesting to occur in a wider variety of areas within a watershed. Logs were once primarily transported by river and are now transported by trucks along constructed roads. Logs used to be removed from the forest by mules and railroad, and these mechanisms have been replaced by tractors and cabling networks.

Forest practices that have resulted in increased fish production or improved habitat have not benefitted coho salmon. Current forest practices in California have been shown to sometimes result in favorable habitat modification, such as increased water yield (Keppeler 1998), increased insect productivity (Hicks et al. 1991), and increased salmonid productivity (Graves and Burns 1970; Nakamoto 1998). The changes are associated with increased numbers of steelhead trout and improved steelhead trout habitat, and most likely, these environmental changes have had a detrimental effect on coho salmon habitat.

Current forestry activities that affect coho salmon habitat include: construction and maintenance of roads and stream crossings; tree felling; yarding felled trees to log landings; removal of stream-side vegetation; site preparation; and post-harvest broadcast burning in harvest units near watercourses. Table 11 describes forestry practices, changes to the landscape, and the potential effects on salmonid habitat conditions. The effect of historical practices is also significant because many now-discontinued practices have been implicated as still impacting salmonid habitat. Significant historical practices include construction and maintenance of splash dams, artificial flooding, removal of trees along the stream-side corridor, removal of in-stream debris, construction of roads and landings, use of equipment adjacent, near, or in streams, and clear-cutting.

Legacy impacts have been documented in northern California: in Mendocino County, Caspar Creek is still recovering from harvesting that occurred in the nineteenth and early twentieth centuries (Ziemer et al. 1991), and the North Fork Garcia River is still recovering from heavy harvesting during the 1950s, 1960s, and 1970s (Monschke 1996). The effects in the Garcia watershed were dramatic, with near elimination of rearing and spawning habitat, and stream flow sometimes becoming subsurface.

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Table 11. Forestry activities and potential changes to stream environment, salmonid habitat, and salmonid biology.¹

Forest Practice	Potential effects to stream environment	Potential effects to salmonid habitat	Potential effects to salmonid biology
timber harvest in the riparian zone	increased incident solar radiation	increased stream temperature, light levels, and primary production	decreased growth efficiency; increased susceptibility to disease; increased food productivity; changes in growth rate and age at smolting
	decreased supply of LWD	decreased cover, storage of gravel and organic debris, and protection from high flows; loss of pool habitat and hydraulic and overall habitat complexity	decreased carrying capacity, spawning gravel, food production, and winter survival; increased susceptibility to predation; loss of species diversity
	increased, short-term input of LWD	increase in number of pools and habitat complexity; creation of debris jams	increased carrying capacity for juveniles and winter survival; barrier to migration and spawning and rearing habitat
	increased influx of slash	increased oxygen demand, organic matter, food, and cover	decreased spawning success; short-term increase in growth
	stream bank erosion	reduced cover and stream depth	increased carrying capacity for fry; decreased carrying capacity for older juveniles; increased predation
		increased in-stream fine sediment; reduced food supply	reduced spawning success; slower growth rates for juveniles
timber harvest on upslope areas	altered stream flow	temporary increase in summer stream flow	temporary increase in survival of juveniles
		increased severity of peak flows during storm season; bedload shifting	increased egg mortality
timber harvest on upslope areas and road construction and use	increased erosion and mass wasting	increased in-stream fine sediment; reduced food supply	reduced spawning success, growth and carrying capacity; increased mortality of eggs and alevins; decreased winter hiding space and side-stream habitat
		increased in-stream coarse sediment	increased <i>or</i> decreased carrying capacity
		increased debris torrents; decreased cover in torrent tracks; increased debris jams	blockage to migration of juveniles and spawning adults; decreased survival in torrent tracks
	increased nutrient runoff	increased primary and secondary production	increased growth rate and summer carrying capacity
	stream crossings	barrier in stream channel; increased sediment input	blockage or restriction to migration; reduced spawning success, carrying capacity and growth; increased winter mortality
Scarification and slash burning	increased nutrient runoff	increased primary and secondary production	temporary increased growth rate and summer carrying capacity
	increased input of fine organic and inorganic sediment	increased sedimentation in spawning gravels and production areas; temporary increase in oxygen demand	decreased spawning success; increased mortality of eggs and alevins

¹ Adapted from Hicks et al. 1991

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Water temperature: Alteration of water temperature regimes is considered one of the most important potential impacts from forest practices (Beschta et al. 1987; Murphy 1995; Ligon et al. 1999). Increased ambient air temperature and solar radiation due to the removal of stream-side vegetation and canopy are the causes of the increased water temperature. The change in solar radiation is the primary agent of increased water temperature, especially the daily temperature peaks. Average water temperature may not show as pronounced a change to solar radiation because the reduced canopy also facilitates re-radiation of heat at night. Increased water temperature, resulting from clear-cutting, has been linked to earlier-than-normal emergence of fry (Holtby 1988), increased growth rates because of greater availability of benthic invertebrates (Holtby 1988; Hicks et al. 1991), significant decreases in the number of emergent fry (Moring 1975; Hall et al. 1987; Hicks et al. 1991), and earlier and smaller out-migrating smolts (Moring 1975; Hall et al. 1987; Holtby 1988, Hicks et al. 1991). Increased water temperature does not have to be lethal to impact coho salmon. Sublethal temperature regimes, both low and high, impact anadromous salmonids and are considered as significant as lethal temperatures because sublethal temperatures impact the growth, physiological processes, and behavior of anadromous salmonids (Bjornn and Reiser 1991; Ligon et al. 1999).

Kopperdahl et al. (1971) studied water quality in logged and unlogged areas of six coastal streams in northern California and found that harvesting had its greatest effect on water temperature. Maximum summer temperatures in streams occurring in unlogged areas were below 15.5° C, while temperatures in logged areas approached 21.1° C (Kopperdahl et al. 1971). Temporal temperature data were short-term, only extending two to three years for any given stream, and temperature changes were not considered to be lethal or sustained long enough to affect growth and physiological maintenance of fish. Clear-cut harvesting and removal of vegetation for roads resulted in the greatest temperature increases, while either alternating clear-cut blocks with uncut blocks, or combining selective tree harvest and maintaining a non-cut stream buffer maintained lower water temperatures. Dorn (1969, as cited in Kopperdahl et al. 1971) found that removal of canopy resulted in as much as a 140% increase in solar radiation and a 11° C rise in water temperature in Caspar Creek in Mendocino County. Hall and Lantz (1969) found that clear-cut harvesting resulted in maximum stream temperatures exceeding the ultimate upper incipient lethal temperature for coho salmon (Brett 1952). Meehan et al. (1969) found that logging on Alaskan coastal streams resulted in a 5° C increase in maximum stream temperature. The extensive clear-cut harvesting that led to these types of temperature changes no longer occurs under current California Forest Practice Rules.

Increased water yield after logging has been hypothesized as a moderating factor for the loss of canopy. Removal of vegetation, and its effects on evapo-transpiration rates, results in greater water yield after timber harvest, and increased water yield and summer flow has been demonstrated in California (Krammes and Burns 1973; Keppeler and Ziemer 1990; Keppeler 1998). Keppeler (1998) found that both clear-cut and selective harvesting resulted in increased water yield, with greater yields from clear-cutting, and that the variation in precipitation has a major role in variation in yields. However, this yield is short-lived ($x < 5$ years) and sporadic (Keppeler and Ziemer 1990; Keppeler 1998). More importantly, the yield did not buffer water temperature (Krammes and Burns 1973), and in fact, moderation of summer water temperature was attributed to stream-side canopy and not increased water yield (Keppeler 1998). Hicks et al. (1991) also concluded that positive effects due to reduced canopy are more than offset by negative changes to thermal regimes.

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Monschke (1996) believed that timber harvest from 1953 to 1988 in the riparian zone of the North Fork Garcia River likely increased water temperature, but that other impacts were far greater and made any increase a moot point. Impacts to LWD, sediment, and stream-side cover were considered to be such that anadromous salmonids were excluded regardless of increases in water temperature. From 1988 to 1994, conservation of the riparian area resulted in the recovery of the stream course canopy, resulting in water temperatures adequate for steelhead trout and coho salmon; but only steelhead trout were observed in 1995 and 1996 .

The relationship of water temperature to anadromous salmonids is not a simple one-to-one function, rather it is a function of temperature, available prey, thermal refugia, acclimation, life stage, and species. In their review of FPR and salmonid habitat, the Scientific Review Panel (SRP) (Ligon et al. 1999) stressed that the site-specific nature of these interactions is key to understanding what water temperature regimes represent the suitable, optimum, lethal, and sublethal for salmonid species that inhabit the watercourse. Temperature regimes both beneficial and deleterious to coho salmon as a species, vary across its northern Pacific range, and the suitable and optimum thermal regimes for northern California coho salmon probably differ from elsewhere within its entire range (Brosofske et al. 1997, as cited in NMFS 2000). SRP concluded that until thermal requirements studies take into account physiological conditions in the wild, site-specific thermal regimes, and the effects on local salmonid populations, the “impacts on salmonids as a result of timber harvesting will remain in the realm of conjecture.” (Ligon et al. 1999). Welsh et al. (2001) also concluded that acquisition of such environmental temperature measures were needed in haste because many native fish and amphibians of the Pacific Northwest are threatened by habitat alteration.

One study showed that changes in water temperature alone did not necessarily lead to detrimental effects on coho salmon. In the Navarro River, Mendocino County, adequate sources of prey and cool-water refugia allowed coho salmon juveniles to survive. The increased prey, necessary because of elevated metabolism brought on by increased water temperature, and the thermal refugia, in the form of deeper pools, were thought to have allowed juvenile coho salmon to survive in waters thought to be physiological stressful and provided for good growth rates and apparent healthy condition (Rich 1991 as cited in Ligon et al. 1999).

Notwithstanding this study and the importance and validity of the SRP’s criticism (that much more work needs to be done in order to understand the relationship of water temperature across the range of northern California anadromous fishes), increases in water temperature have been shown to affect coho salmon (Beschta et al. 1987; Sandercock 1991) and elevated water temperature has been demonstrated to be a good, if not the best, indicator for the absence of juvenile coho salmon in timber harvested watersheds in northern California (Ambrose et al. 1996; Ambrose and Hines 1997, 1998; Hines and Ambrose nd; Welsh et al. 2001). Ambrose et al. (1996), Ambrose and Hines (1997, 1998), and Hines and Ambrose (nd) studied water temperature and presence/absence of juvenile coho salmon, considered the most sensitive life-stage to water temperature, on Ten Mile River. Welsh et al. (2001) studied the same phenomenon in the Mattole River watershed, actually gathering data from 21 tributaries in the watershed. In each study, the researchers found that maximum temperature measures were good predictors of the presence or absence of juvenile coho salmon.

Ambrose et al. (1996) and Ambrose and Hines (1997, 1998) found that maximum weekly average temperature (MWAT) was a reliable predictor of coho salmon presence and that MWAT

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of 18.0-18.3EC was the critical measure. Hines and Ambrose (ND) refined the findings to state that a MWAT of 17.6EC was the best fitting MWAT value for predicting the presence or absence of the species. More importantly, they found that the number of days exceeding the MWAT, and not the value itself, was the defining point beyond which juvenile fish ceased to be present. They calculated four MWATs, ranging from 15.9EC to 18.3EC, that could be used to indicate the presence or absence of fish. Another important point these authors made was that persistence of juvenile fish “does not imply health or success”, and rather, that temperature thresholds provide for a “reasonable way to rule out unacceptable temperature conditions”. Welsh et al. (2001) measured MWAT and also measured maximum weekly maximum temperature (MWMT) to determine if either variable modeled the presence or absence of juvenile coho salmon. Both measurements correctly predicted the absence of the fish in 18 out of 21 streams sampled. Streams containing juvenile coho salmon had MWMT and MWAT of 18.0EC and 16.7EC or less, respectively. Moreover, all streams with MWMT less than 16.3EC and MWAT less than 14.5EC had coho salmon present.

The data gathered thus far for California suggests coho salmon prefer cool waters, avoiding water above 14.5-17.6EC, and that timber harvest practices have negatively altered water temperature regimes, considered the most important habitat attribute for coho salmon. In some cases, local fish populations have survived or have at least persisted, but increases in water temperature have not translated in favorable conditions for the species. There are findings from water temperature field studies in northern California that may provide a basis for establishing maximum temperature thresholds for coho salmon along the north coast, and that these thresholds can be used to guide timber harvest practices to ensure essential water temperature regimes are maintained. Such an effort would not be unprecedented. Work done by Ambrose et al. (1996), Ambrose and Hines (1997, 1998), Hines and Ambrose (ND), Valentine (1994, 1996), and CDF (1998) was conducted, in part, to analyze water temperature suitability for salmonids on private and state forest lands.

Sedimentation: Forest practices increase sediment delivery to watercourses, resulting in increased fine and coarse sediment loads in streams and increased water turbidity (Furniss et al. 1991; Murphy 1995; Spence et al. 1996; Ligon et al. 1999). In California, the short-term and long-term effects to coho salmon and their habitat are complex. It is also acknowledged that sediment from poorly constructed roads and harvesting on unstable slopes is having legacy and current impacts in coastal California watersheds (Ligon et al. 1999). Increased sedimentation is a consequence of increased bare soil and disturbed ground from removal and yarding of trees; the design, construction, use, and maintenance of roads; and landslides associated with harvested areas. Increased sedimentation and mass-wasting has been linked to logging in western North American (Megahan 1972; Chamberlin et al. 1991; Furniss et al. 1991; Corner et al. 1996; Spence et al. 1996) and specifically in coastal northern California (Burns 1970; Monschke 1996; Cafferata and Spittler 1998). Such changes are deleterious to spawning bed composition, egg incubation, fry survival, and juvenile growth and productivity (Ligon et al. 1999). Silt from poorly harvested areas and/or improper road construction can clog spawning gravels, suffocating eggs or alevins (Cordone and Kelly 1961). Sediment levels of greater than 40 g/L cause distal deterioration of gill filaments (Lake and Hinch 1999). Studying salmonids in a laboratory, Cordone and Kelly (1961) found that mortality occurred at 100 g/L. Though this concentration is approximately an order of magnitude higher than in natural salmonid streams, they noted that natural fluvial suspended sediments at much lower concentrations caused stress and mortality.

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Burns (1970) investigated two species of trout and coho salmon in coastal streams and watersheds in Mendocino and Humboldt counties and found that most of the impacts were due to forest practices. Cafferata and Spittler (1998) specifically investigated differences in sedimentation that could be associated with old and new forestry practices, and they found that new practices resulted in a 75% reduction in erosion. Monschke (1996) also studied changes in sedimentation related to changes in forestry practices and concluded that selection harvesting conducted since the late 1980s did not contribute measurably to sedimentation, and staying out of the riparian area in conjunction with better road management was resulting in slow recovery from past, high levels of sediment.

Sediment from road construction, use, improper maintenance, crossings, and failures are sources of increased sedimentation in watercourses (McCashion and Rice 1983; Furniss et al. 1991; Murphy 1995). Road design, construction, maintenance, and use affect several stages of salmon, including migrating and spawning adults, eggs, fry, and rearing juveniles (Furniss et al. 1991). In four counties of northern California, McCashion and Rice (1983) found that roads, while the cause for only 20% of the total number of landslides, were responsible for 56% of the erosion, compared to natural slides being responsible for 80% of the total slides but only 44% of the erosion. This research also demonstrated two other significant aspects of road-erosion relationships. First, different types of roads contributed differently to the amount of erosion; seasonal roads, followed by main haul roads and secondary roads, produced the highest rates of erosion. Second, 38% of the erosion could have been prevented by improved road construction and maintenance. Monschke (1996) also concluded that poor road management was an important factor in excess erosion and landsliding and that changes in road management, due to the Z'Berg-Nejedly Forest Practice Act (FPA) of 1973, greatly improved this. However, he also found that both the quality and quantity of sediment transported to streams was impacting salmonids.

Timber harvest can result in increased suspended sediment, sediment deposition in pools and gravel, and reduced gravel permeability (Moring et al. 1994; Ligon et al. 1999). Suspended sediment can impact all life-stages, but especially juveniles and fry. Changes to the gravel conditions impact the survival to emergence of egg and alevin life-stages directly, and thus, reduce the spawning success of adults. There is some evidence that suspended sediment may return to pre-harvest concentrations within seven years (Moring et al. 1994) but this would be in the absence of stressing storm events or additional ground-disturbing activities subsequent to timber harvest. Krammes and Burns (1973) found that road construction increased suspended sediments in Caspar Creek watersheds and that levels were four times that of pre-construction concentrations after the first storm flows. However, they also found that turbidity decreased quickly and, although higher than pre-construction levels, was "not excessive."

Forestry practices are also tied to changes in coarse sediment transport. Monschke (1996) found that debris torrents and slides occurred because of harvesting and flooding events in the North Fork Garcia River between 1953 and 1988. Slides delivered considerable quantities of sediment and woody debris. The LWD contributed to debris jams that became effective migration barriers. Many of the torrents and slides were associated with roads, and recovery from the sedimentation is only now occurring.

Altered stream flow: Increased peak and storm flows due to road construction has also been considered a potential impact to anadromous salmonids. Such flows would both transport

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more sediment and alter the annual and seasonal hydrologic regimes of the watercourses. These linkages have been noted in other western states such as Washington (Cederholm and Reid 1987). The change in hydrology is usually directly proportional to the size of the watershed and the size of the area harvested and is also a function of the type of timber harvest activity, precipitation, geology, and soil type. Less intensive harvest practices, such as thinning and selective harvesting, results in less effect on water yield and altered stream flow. Disturbance during the wet season, or on more sensitive soil or unstable slopes, results in greater impacts. Considerable effects have been shown in British Columbia and the more northwestern states. However, in California, road construction and timber harvest has not been shown to effect large peak flows or change major, channel-forming flows (Ziemer 1981; Wright et al. 1990). In Caspar Creek, Mendocino County, low to moderate flow changes resulting from timber harvest have not shown to alter net bed-load transport (Lisle and Napolitano 1998).

Timber harvest also alters summer flow. The greatest increases have been documented in the Oregon Cascades (Spence et al. 1996), but in California, little research has been conducted to understand the changes or duration to summer flow or their effect on anadromous salmonids. Keppeler and Ziemer (1990) studied summer water yield increase on Caspar Creek, Mendocino County, and found that the increase was slight and disappeared five years after timber harvest ceased.

Large woody debris: Impact to LWD is considered to be another of the more important, potential impacts of forestry practices to anadromous salmonid freshwater habitat (Hicks et al. 1991; Murphy 1995). The potential results of harvesting activities include decrease of both in-stream and streambank LWD, and a decreased future supply of LWD from upslope sources. The role of LWD, the relationship of forestry practices, and rectifying existing depressed levels of in-stream LWD has been studied along the west coast (Bryant 1983), in Washington (Cederholm et al. 1997), and in Alaska (Lisle 1986), and specifically for coho salmon (Bryant 1983; Lisle 1986; Cederholm et al. 1997). In California, timber harvest has been shown to cause a short-term, greater contribution of LWD to streams, which resulted in increased number of pools and stream habitat complexity (Lisle and Napolitano 1998; Napolitano 1998). However, coho salmon did not respond positively to these improved habitat conditions, and Lisle and Napolitano (1998) theorized that the removal of the source of future LWD would result in a greater departure from the natural volume of LWD, decreased sediment storage capacity, a decrease in the number of pools, and an overall simplification of stream habitat. Monschke (1996) also concluded that a decrease in future recruitment of LWD would be an issue in the North Fork Garcia River, but he found that the short-term, greater supply of LWD due to harvesting was a habitat impact, not a benefit, that resulted in debris barriers to salmonid spawning and rearing areas.

The SRP summarized the impacts associated with decreased LWD due to forestry activities in California, and these impacts include loss and reduced complexity of pool habitat, reduced carrying capacity for juvenile fish, reduction in backwater and stream margin habitat important to emergent fry, more simple and less stable stream channels, reduction in refugia from high-velocity flows, reduced retention of spawning gravel, and loss of sediment important for macro-invertebrate prey (Ligon et al. 1999).

Historically, timber operations included removal of stream-side vegetation and in-stream woody debris to facilitate transport of logs via waterways. When harvesting moved upstream to streams too small to facilitate ready transport of logs, splash dams were built to hold logs and

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water until there was sufficient water to allow the logs to be sluiced downstream (Hicks et al. 1991). Once land routes became the primary means of transporting harvested materials, extensive removal of in-stream LWD and the use of splash dams effectively halted. In the 1950s and 1960s, and under the encouragement and recommendation of the Department, the practice of removing LWD from stream channels became common. Removal of LWD was done primarily to facilitate the removal of remnants of splash dams, but also because there was a belief that such removal would generally benefit anadromous salmonids and trout by decreasing the magnitude and frequency of migration barriers (Flossi et al. 1998). In hindsight, the former rationale was sound, while the latter was not. Research in western North America has demonstrated that removal of LWD impacts habitat and salmon populations (Hicks et al. 1991; Ligon et al. 1999).

Harvesting itself effects the future recruitment of LWD. Conversion of old-growth redwood forests to either younger second-growth forests and predominantly hardwood communities has resulted in smaller woody debris that decay more rapidly and provide less channel stability and salmonid habitat than conifer LWD (Bragg and Kershner 1999, as cited in Ligon et al. 1999; Ligon et al. 1999). Recruitment of LWD has been shown to be an issue in harvested watersheds of northern California. Lisle and Napolitano (1998) advanced the idea that short-term increased LWD would be followed by decades of decreased LWD supply due to logging. Monschke (1996) found that riparian recovery was dominated by alders (*Alnus* spp.), contributing little to the supply of high-quality LWD, and that recovery of conifers was going to take much more time.

Stream-side vegetation and canopy cover: Removal and alteration of vegetation along and near watercourses impacts salmonids (Cederholm and Reid 1987), and studies show that protection of stream-side vegetation benefits trout and anadromous salmonids, including coho salmon (Burns 1972; Hicks et al. 1991). Harvest of canopy-creating trees from stream-side habitat affects cover from predation, water temperature, the watershed's ability to absorb precipitation, water flow timing, erosion, bank stability, retention of in-stream woody debris, recruitment of LWD, and habitat complexity (Murphy et al. 1986; Meehan 1991; Moring et al. 1994; Monschke 1996). Removal of near-stream vegetation can result in increased water temperature, both short- and long-term (Moring et al. 1994). Prior to the changes in forest practices, clear-cutting of stream-side vegetation was shown to increase water temperature of salmon-bearing streams (Brett 1952; Kopperdahl et al. 1971; Hall et al. 1987, Holtby 1988). Upstream monitoring also has shown that clear-cutting increased sedimentation (Corner et al. 1996).

Monschke (1996) found many effects from the result of harvesting in the riparian areas of the North Fork Garcia River. Those effects included substantial increases in contiguous open river reaches, the length of canopy gaps, and stream width. Erosion and sedimentation resulting from exposed and disturbed soil resulted in sediment deposits that destroyed stream-side vegetation and entrainment of LWD, which, in turn resulted in knock-down of additional stream-side vegetation.

Physical barriers: Certain forestry activities can result in barriers that block or impede adult and juvenile fish movement. These activities include construction and maintenance of road crossings, debris dams, and debris jams. Road crossings, especially those using culverts, are elements of both historical and contemporary practices, and these crossings, unless properly designed and maintained, prevent fish passage. Historical timber harvest often resulted in debris

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dams, the larger of which became physical barriers to upstream spawning runs and downstream out-migration of juveniles. Although debris dams have been removed or targeted for modification since the 1960s and current forestry practices pay particular attention to allowing fish passage through road crossings, historical barriers may have caused fish to cease use of habitat upstream of barriers. The SRP advised that attention be given to barriers to habitat historically accessible to anadromous salmonids (Ligon et al. 1999). Debris jams are a result of excess LWD and slash being transported into stream channels and accumulating to the point of preventing anadromous fishes from passing. Monschke (1996) tied timber harvesting to such debris jams on the North Fork Garcia River.

Dissolved oxygen by life stage: Adverse changes to DO levels following timber activities vary in degree, and the effects due to changes in DO are influenced by interstitial flow, water temperature, and stream productivity. Hicks et al. (1991) summarized the effects to DO in small harvested streams in Oregon and found reduced levels of DO below that which is suitable for survival and growth, but that major changes to DO in surface water was not likely. More importantly, they found DO decreased in redds, where it is crucial for egg and alevin survival and development, and that reduced DO might impact juvenile size, viability, and fitness. They also believed that egg or alevin mortality from reduced DO was rare. Moring et al. (1994) also summarized several studies and found that logging resulted in decreases in DO levels that would threaten continued survival and growth of salmonids and stated the DO levels were “reaching critical values.” The few studies done in California streams inhabited by coho salmon found that timber operations either did not effect DO (Krammes and Burns 1973) or were not outside the normal DO range (Kopperdahl et al. 1971).

Effects on estuaries: Forestry activities can affect coastal estuaries inhabited by coho salmon. Some effects originate from activities within the estuary themselves, while others are a result of forestry activities upstream from the estuary. Within the estuary, forestry practices of storage, log handling, and transformation has taken place for over 100 years. In California, unlike other areas such as Alaska or British Columbia, these practices are much reduced from historical levels. The primary concern regarding upstream influence of forestry practices is sediment transport and filling-in of estuaries. Though this has obviously occurred in California, there is little documentation. Redwood Creek’s estuary has been affected, and the Mattole River estuary has also likely been impacted, but the degree is unknown (Steve Cannata, pers. comm.). There is less information on other estuaries within the range of northern California coho salmon. Consequently, the historical, cumulative, or current effects of forestry practices to estuaries and estuary habitat of coho salmon are not known.

Conclusions: Current forestry activities with the greatest impact on coho salmon appear to be the construction and maintenance of roads (especially when those roads are poorly designed, substandard, or not properly maintained) and timber harvest along the stream-side or on unstable slopes. These and other activities result in increased sedimentation, decreased LWD, increased water temperature, and decreased cover, and appear to be important. However, current information suggests that changes to DO, stream flow, and water turbidity resulting from forestry activities are ephemeral or negligible and have not been documented to impact northern California coho salmon and their habitat.

The Department’s conclusion is that historical forestry practices impacted watersheds inhabited by northern California coho salmon, and that current activities (e.g. road construction,

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use, and maintenance; activity near streams and on unstable slopes; removal of sources of future LWD) still affect important habitat elements essential to every life-stage of coho salmon that inhabit coastal streams and rivers.

Water Diversions and Fish Screens

A substantial amount of coho salmon habitat has been lost or degraded, due primarily to decreased flows because of water diversions and groundwater extraction, and unscreened or poorly screened diversions that entrain juvenile fish. Losses of coho salmon result from a wide range of conditions related to unscreened water diversions and substandard fish screens. Fish are entrained into the diversion channel and removed from their natural habitat. They are also killed or injured by diversion pumps or stranded in irrigation canals.

Primary concerns and considerations for fish at diversions that are unscreened or equipped with poorly functioning screens are:

- fish passage upstream (for adults) of all fish species;
- overall survival of downstream migrants;
- exposure time of downstream migrating juveniles to the face of the screen;
- screen bypass flow, which is a function of approach (perpendicular into the screen) and sweeping (parallel along the screen) velocities for fish moving in front of the screen;
- entrainment of juvenile fish into the diversion;
- impingement of juvenile fish on the screen due to high approach velocities in front of and/or low sweeping velocities past the screens;
- sediment accumulation patterns behind and in front of the screens, which modifies approach velocity, sweeping velocity, and predation patterns;
- predator holding areas that could be created by localized hydraulic effects of the fish screen and related facilities;
- entrapment of juvenile fish in eddies or other hydraulic anomalies where predation can occur;
- elevated predation levels due to concentrating juveniles at diversion structures;
- disruption of normal fish schooling behavior caused by diversion operations, fish screen facilities, or channel modifications; and
- loss of habitat due to decreased flows and water.

Instream Flows

Introduction: Large dams often alter the natural hydrograph that anadromous fish have evolved with and can preclude recruitment of spawning gravels from upstream sources to areas below the dam. Water impounded by dams can reduce the frequency and magnitude of flows necessary to transport sediment, allowing fines to accumulate and armoring spawning gravels below the dam. Dams alter flow regimes in downstream reaches, commonly reducing flood peaks (at least for moderate floods). Dams can also alter the temperature regime downstream of the dam, making the stream cooler or warmer depending upon whether the releases from the reservoir are hypolimnetic or epilimnetic. Base flow during dry months may be either decreased or increased, depending upon reservoir operation and whether water is diverted directly from the reservoir or from the channel downstream.

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Smaller diversion structures can have similar impacts to anadromous fish, but generally these are reduced in scale and are more localized. In some streams, impoundments created by diversion structures can create conditions lethal for young salmonids. During late spring and summer months, water quality and temperature conditions can deteriorate making these areas unsuitable for juvenile rearing (CDFG 1997a). In nutrient rich waters, impoundments can create conditions favorable for aquatic plant growth and areas of increased organic decay and elevated aerobic bacterial activity. Wide ranging DO levels resulting from these conditions can be lethal to rearing salmonids.

Diversions for stockwater, domestic, and municipal purposes usually occur year-round while agricultural diversions are generally seasonal in nature (mid-spring to mid-fall). In some cases, agricultural diversions may continue into the winter to recharge water storage facilities that are used later for irrigation. Naturally low water conditions, such as that which typically occurs during the spring, summer, and early fall can be unfavorable for salmonids, however, these problems can be greatly exacerbated by agricultural water diversions (KRBFTF 1991). Resultant flows below diversion points decrease the amount of physical space available to juvenile salmonids. Water temperatures can change more dramatically, both diurnally and seasonally, due to decreased depth and reduced water volume. Agricultural return water can be higher in temperature and nutrient content than the receiving waters, further eroding water quality and habitat availability (CDFG 1997a).

In many rivers, summer and fall baseflow periods are particularly critical for survival of fish and other aquatic organisms. Reduction in these baseflows can have severe ecological impacts. In decreasing order of severity, these effects of reduced baseflows can range from completely drying the channel and lowering the water table (desiccating aquatic and hyporheic organisms, and potentially dewatering riparian vegetation), to drying shallower parts of the channel while maintaining isolated pools (eliminating connectivity of surface waters, increasing predation by terrestrial animals, and reducing water quality), or to reducing the flow and velocity (causing water temperatures to rise and/or DO levels to drop) (Kondolf et al. 1990).

Depletion and storage of natural flows can drastically alter natural hydrological cycles and create significant impacts to downstream reaches by: increasing juvenile and adult mortality due to migration delays resulting from insufficient flows or habitat blockages; reducing habitat due to deterring and blockage; stranding of fish due to rapid flow fluctuations; and increasing mortality due to higher water temperatures (CACSSST 1988; CDFG 1991; Berggren and Filardo 1993; Reynolds et al. 1993; Chapman et al. 1994; Cramer et al. 1995; NMFS 1996). In addition to these factors, reduced flows negatively affect fish habitats due to increased deposition of fine sediments in spawning gravels, decreased recruitment of LWD and spawning gravels, and encroachment of riparian and non-endemic vegetation into spawning and rearing areas resulting in reduced available habitat (CACSSST 1988; FEMAT 1993; Botkin et al. 1995; NMFS 1996).

The following is a description of impacts associated with instream flows in several river basins within the range of coho salmon in California. These impacts have been shown to effect salmonids in general but likely impact coho salmon.

Klamath River: Anadromous fish have been blocked from the upper Klamath River watershed since 1918 when Copco #1 Dam was constructed. Iron Gate Dam, constructed in

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1962, re-regulates peaking flows generated by upstream facilities and is the present upper limit of anadromous fish distribution in the Klamath River.

Federal Energy Regulatory Commission (FERC) minimum flows at Iron Gate Dam have frequently not been met during the period 1961 to 2000 due to the fact that the United States Bureau of Reclamation's (USBR) Klamath Project controls most of the flow in the Klamath River. In the past, the water project has provided water to irrigation at the expense of downstream deliveries during below average water years. This situation is especially pronounced during droughts. For example, the monthly mean streamflow below Iron Gate Dam did not meet FERC minimum standards from February 1991 through February 1993, a period of 25 consecutive months.

Existing flows in the Klamath River below the Scott River confluence during the summer period can result in lethal combinations of high temperature and low DO, as evidenced by fish kills. Temperatures can reach a high of 80°F for up to 10 days each year. However, cold water refugia, especially at the mouths of a number of tributaries, are well documented and help ameliorate the effects of thermal stress (Bartholow 1995).

Long-term changes to flow patterns resulting from water impoundments and diversions can have huge impacts on anadromous fish. Historically, the Klamath River and its Salmon, Scott and Shasta river tributaries supported significant populations of spring-run chinook salmon (Snyder 1931). Today, spring-run chinook are considered extinct in the Klamath system upstream from the Salmon River due, in part, to inadequate summer flow conditions, which eliminated the deep, cool pools they require to over-summer (KRBFTF 1991). In the Shasta and Scott rivers, low flow conditions can impact the timing and distribution of adult salmon spawners. During dry years, the main run of adult chinook salmon into the Shasta River is delayed until October 1, which is the end of the summer irrigation season (CDFG 1997a). In wet years, salmon have access to over 38 miles of stream, but may only access 10-15 miles in dry years (CH2M Hill 1985). In some years, returning chinook spawners are unable to leave the canyon section of the Scott River to migrate to upstream spawning areas because of low flows (Mark Pisano, pers. comm.). Diversions of as little as 10-15 cfs for stock watering can be critical to migration access when the Scott River is only flowing at 35 cfs (KRBFTF 1991).

Diversions for agriculture occur in several other, smaller tributaries to the middle Klamath River. These are; Willow Creek, Little Bogus Creek, Bogus Creek, Horse Creek, Cottonwood Creek, and Grider Creek (KRBFTF 1991).

Shasta River: The Shasta River watershed consists of approximately 507,000 acres (793 mi²) of which about 28 percent (141,000 acres) is irrigable and exists primarily below Dwinnell Dam (CDWR 1964). The Shasta River was dammed at RM 37 to form Dwinnell Reservoir (Lake Shastina) in 1928. In 1955, the height of the dam was raised, which increased the total storage capacity to 50,000 acre-feet. Seven major diversion dams and several smaller dams or weirs exist on the Shasta River below Dwinnell Dam. Numerous diversions and associated dams exist on other major tributaries as well, including Big Springs Creek, Little Shasta River, and Parks Creek. When all diversions are operating, flows are substantially reduced and, in the case of the Little Shasta River, stream flows cease entirely in the lower several miles of stream during the summer and fall period. There are over 100 known water diversions within the Shasta River watershed (State Water Resources Control Board 1996).

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Agricultural return water is often considerably warmer when it flows back into the river. This runoff may be rich in organic matter, which can raise nitrogen and phosphorus levels in parts of the river. Diversion dams slow the river's flow, which allows the water to warm in the summer. The dams also create a pond-like environment, rich in nutrients, where algae bloom in abundance. This can cause the water to become super-saturated with oxygen during the day and cause oxygen depletion at night.

The onset of the irrigation season in the Shasta River watershed in some years can have a dramatic impact on discharge if large numbers of irrigators begin taking water simultaneously. This can result in a rapid decrease in flows below the diversions leaving fish stranded in shallow pools and side channels (CDFG 1997a). In some instances, channels can become entirely de-watered (KRBFTF 1991).

Scott River: The unstable granitic soils and past human activities (e.g. logging, roads) along the west side of Scott Valley have been contributing to the Scott River's problem of excessive fine sediment. This fine sediment comprises a large percentage of the Scott River's substrate. This, along with the relatively large amount of water diverted from the Scott River and its tributaries, has resulted in reduced river flows and relatively high annual water temperatures. Because of these water quality problems, the Scott River has been listed as an "impaired" waterway under section 303(d) of the CWA. A total maximum daily load (TMDL) plan will provide the method for assessing the environmental problems that resulted in the "impaired" listing of the Scott River and will develop a strategy to reach acceptable water quality standards within a set time frame. California's Regional Water Quality Control Board (RWQCB) for the Scott River region will establish TMDLs by the year 2005.

Agriculture is the single largest water user within the Scott Valley. It has been estimated that gross water use for agriculture is 98,100 acre-feet and net use is 78,000 acre-feet (taking into account evapo-transpiration and ditch loss). Most of the irrigation diversions on the Scott River operate from April 1 through October 15 pursuant to the 1980 Scott River Adjudication decree of the Superior Court of Siskiyou County. This decree recognizes 680 total water diversions, which cumulatively could divert 894 cfs from the Scott River and its tributaries (CH2M-Hill, 1985). Earlier adjudication decrees allocated water for irrigation, stock-water and domestic use from the Shackleford/Mill Creek drainage in 1950, and from the French Creek drainage in 1958. Previous riparian, pre-1914 claims, and appropriative water rights were included in all of the court adjudicated decrees within the Scott watershed.

Diversions from streams for both stockwater and domestic use were also allocated under these court adjudicated decrees. Many domestic users are scattered throughout the valley and foothills of the Scott watershed, most of these utilizing ground water from individual wells for their household and landscaping needs. Information on local residential and commercial water use is sparse.

Within the past six to 10 years, improvements in some city water delivery systems and the metering of users within some local municipalities have significantly reduced municipal and domestic usage. In 1990, the average domestic water use within Etna and Fort Jones, the two largest municipalities, was 266 and 170 gallons per person per day, respectively. The City of Etna diverts water directly from Etna Creek while Fort Jones pumps water from the underflow of Moffett Creek and the Scott River. Assuming an average local water demand of 200 gallons per

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person per day, the total urban (i.e., domestic/residential/municipal) water use in 1990 was estimated to be 1,800 acre-feet (SRWCRMP 1995). Stockwater use is estimated to be 504 acre-feet based on an estimated maximum 30,000 head of cattle within the Scott River watershed utilizing an average of 15 gallons per day. The gross use taken under a stock-water right, including ditch loss, is not known but is judged to be quite high in some instances.

In most years, low flows in the Scott River occur during the months of June to November in the mainstem and in some major tributaries. During periods of drought, large portions of the mainstem Scott River are completely dry (SRWCRMP 1997). Many thousands of juvenile salmon and steelhead are stranded in some years due to dewatering of streams in the Scott River Basin (SRWCRMP 1997). Stream flows usually go subsurface in the lower reaches of Etna, Patterson, Kidder (including Big Slough), Moffett and Shackleford creeks each summer through early fall. Redds are also sometimes dewatered in the fall when water levels rise and then subside as a result of rainfall patterns in conjunction with diversions (SRWCRMP 1997).

Trinity River: The anadromous portion of the Trinity River extends 112 RM starting at the confluence with the Klamath River at Weitchpec to the upstream limit of fish passage at Lewiston Dam. The major tributaries in this reach are the South Fork Trinity River, New River, French Creek, North Fork Trinity River, and Canyon Creek. The South Fork Trinity River is the largest sub-basin within the Trinity watershed. The Trinity River is impounded at Lewiston and Trinity dams, both operated by the USBR. The former serves as the main water storage facility and the latter as a control facility for regulating releases both in-stream and for export. Trinity Lake and Lewiston Reservoir have storage capacities of 2,448,000 and 14,660 acre-feet, respectively.

Temperatures in the upper 40 miles (64.4 kilometers) rarely exceed 70°F, due to cold releases from Lewiston Dam. However, lower river stream temperatures typically exceed 70°F during the summer months of late July through early September. The Trinity River has been classified by the United States Environmental Protection Agency (USEPA) as sediment impaired. This is partially due to the lack of unregulated flows required to mobilize and transport sediments.

Mad River: Ruth Dam (a.k.a. Robert W. Matthews Dam) was built in 1961 about 80 miles upstream of the mouth of the Mad River in Trinity County to provide water for industrial use (e.g. pulp mills), domestic use, and hydroelectric power. It is a barrier to adult salmonids, and has a considerable influence on streamflow for 80 miles (129 kilometers) downstream of the dam (CDFG 2001b). The Humboldt Bay Municipal Water District operates five Ranney collector wells in the lower portion of the Mad River that have a design capacity of 75 million gallons per day to supply drinking water to Eureka, McKinleyville, Blue Lake, Freshwater, Arcata, and other smaller surrounding communities (CDFG 2001b).

Eel River: Mainstem Eel River flows have been regulated and managed for hydroelectric power and exported for agriculture since 1922. There are two dams associated with the Potter Valley Hydroelectric Project located on the upper mainstem Eel River. Scott Dam forms Lake Pillsbury and Cape Horn Dam forms the Van Arsdale diversion reservoir. A diversion tunnel draws water from Van Arsdale reservoir through a mountain and delivers the water to the Potter Valley Powerhouse. Some of the diverted water is used in Potter Valley. The remainder is

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stored in Lake Mendocino and released to the Russian River where it is used for frost protection and irrigation of crops and other purposes (CDFG 2001b).

Regulated flow releases from Lake Pillsbury change the temperature regime between Scott and Cape Horn dams. Water temperatures become cooler in summer and warmer in winter. The change in water temperature enhances summer rearing for steelhead trout, but can delay juvenile chinook salmon emigration. The delay results in juvenile chinook salmon encountering marginal or lethal water temperatures as they migrate through downstream reaches of the Eel River towards the ocean. Over half the mainstem and tributary channels can be considered thermally lethal during some portion of the summer. There are two additional small hydroelectric facilities on the mainstem Eel River. One is located on Mud Creek (Dobbys Creek tributary) and another on Kekawaka Creek (CDFG 1997b).

Two other reservoirs, Centennial and Morris, are located on Davis Creek, a tributary to Outlet Creek. These reservoirs provide water to the city of Willits. Lake Emily and Lake Adarose are located on Willits Creek, also a tributary to Outlet Creek. These reservoirs provide water to the community of Brooktrails. Benbow Lake is located on the South Fork Eel and is a seasonal impoundment closed only during the summer months. It is currently under review by NMFS to investigate the impact to salmonids.

There are approximately 260 licensed, permitted, or pending water rights within the Eel River watershed. This number does not include riparian users and other diversions that are not registered with the Division of Water Rights of the State Water Resources Control Board (SWRCB).

Russian River: In 1908, the flows of the Russian River were augmented by Eel River water diverted through the Potter Valley Tunnel to generate electricity. In 1922, Scott Dam on the Eel River was constructed to provide a more reliable year-round supply of water to the Potter Valley Powerhouse. Augmentation of the Russian River, via its East Branch, from the Potter Valley Project averaged 159,000 acre-feet per year (an average flow of approximately 21 cfs) between 1922 and 1992.

Coyote Dam, forming Lake Mendocino on the East Branch near Ukiah, was completed in 1959 to provide flood protection and store water for domestic use. Coyote Dam has no fish passage facility. During the winter the dam reduces storm flow peaks and extends the storm hydrograph. During the summer months water is released from Coyote Dam to maintain higher than natural flows for fish, recreation, and redirection to agricultural and domestic consumers. Summer minimum flow releases in the Russian River between the East Branch and Dry Creek is 150 cfs. As a result, summer flows in Healdsburg are some 15 to 20 times greater than the unimpaired flows would be.

Warm Springs Dam, forming Lake Sonoma on Dry Creek near Healdsburg, was completed in 1984 and, similar to Coyote Dam, provides flood protection and water storage for domestic use. During the summer months water is released to Dry Creek, then to the Russian River for redirection at Wohler and Mirabel as domestic water. Summer minimum flow release in Dry Creek is 75 cfs, although it is often significantly greater. The unimpaired summer flow in Dry Creek is significantly less.

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In addition, there are five smaller impoundments on the mainstem Russian River, and approximately 500 licenced or permitted dams on the tributaries to the Russian River (SEC 1996). These diversions range from very small domestic use diversions to large agricultural diversions. The most significant impacts to fishery resources may be caused by frost protection diversions which can take large volumes of water in a short period of time. Irrigation diversions may individually take a much smaller volume of water than a frost protection diversion, but some irrigation diversions are still large relative to the flow of many tributary streams during the summer. In some areas the cumulative impact of several irrigation diversions may be significant. Reduced vegetation cover in much of the Russian River watershed appears to have increased the rate of run-off in the winter and reduced the flow of streams in the summer.

Walker Creek: In the late 1970s following the 1976-1977 drought, a dam was constructed on Arroyo Sausal just upstream of its confluence with Walker Creek to augment the domestic water supply of the Marin Municipal Water District. This dam, which forms the 10,000 acre-foot SoulaJule Reservoir, was constructed just downstream of an older, small agricultural dam. Since its construction, the water from SoulaJule Reservoir has never been used except for periodic test pumpings. To be used, the water from SoulaJule Reservoir must be pumped a considerable distance to Nicasio Reservoir to the south, and from there to the water treatment plant in San Geronimo. Because of this, there has been no significant impact on the total flow of water in Walker Creek. There are, however, some reductions in peak flows, changes in the shape of the storm hydrographs, and increases in summer flow. There may also be some increase in summer water temperature in the area near the dam. Summer flow releases are intended to improve salmon and steelhead trout habitat in most years. Natural surface flows would often cease in many areas by midsummer.

The release of warm reservoir water and higher than natural flows could result in water temperatures above the optimum for coho salmon. Water temperatures recorded throughout the summer of 1998 in the three miles below the dam regularly reached highs between 68⁰ and 74⁰ F. Temperatures in the canyon near the confluence of Chileno Creek, where coho salmon are more likely to be found, would likely be significantly less.

Lagunitas Creek: Lagunitas Creek once supported large numbers of coho salmon and steelhead trout, but populations were significantly reduced by inadequate instream flows, prolonged drought, and habitat loss. There are five large reservoirs in the Lagunitas Creek system: Lagunitas, Bon Tempe, Kent, and Alpine reservoirs on the mainstream and Nicasio Reservoir on Nicasio Creek. Dams forming these reservoirs block access to about 25 percent of the habitat once used by coho salmon. These reservoirs are operated by the Marin Municipal Water District to provide domestic water to the heavily populated eastern Marin County. A small diversion on lower Lagunitas Creek is also used by the North Marin Water District to serve 1,000 to 1,500 residents in the Point Reyes Station area.

Artificial Barriers

Artificial structures on streams fragment aquatic ecosystems by blocking or impeding migration and altering nutrient cycling patterns, streamflows, sediment transport, channel morphology, and species composition. This reduces available habitat and changes habitat conditions for anadromous salmonids and reduces native biodiversity. Stream ecosystem fragmentation occurs when the river or stream continuum is disrupted by barriers such as road

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crossings, dams, severe pollution, or other land management practices, or when surface or subsurface hydrologic connections are severed between the stream channel and adjacent wetlands.

Structures that can potentially block or reduce fish passage include dams, road crossings (bridges, culverts, or low water fords), flood control facilities (concrete channels), erosion control structures (energy dissipaters), canal and pipeline crossings, pits from gravel mining, and other more unique situations. Instream structures have the potential to, depending on conditions, either entirely or partially block fish from accessing upstream reaches and block critical habitat necessary for survival.

Complete blocking occurs not only at large dams, but can also occur at small structures less than 10 feet high if there is not enough streamflow, if the downstream face or footing of the dam slopes away (making the horizontal distance too great to overcome), or if the jump-pool at the foot of the structure is too shallow or non-existent.

Even if stream barriers are eventually negotiated by fish, excess energy expended may result in their death prior to spawning, or reductions in viability of eggs and offspring. At temporal barriers, the delay imposed by one or more stream crossings can limit the distance adult fish are able to migrate upstream before spawning, which can result in under-utilization of upstream habitat and superimposition of redds in lower stream reaches. Migrating adults and juveniles concentrated below barriers with impassable culverts are also more vulnerable to predation by a variety of avian and mammalian species, as well as illegal harvest by humans.

While the upstream movement of adult salmon and the downstream movement of salmon smolts are familiar phenomena, other occasions of fish migration or movement are not generally as well known. Juvenile salmonids move both upstream and downstream in response to various environmental factors. These factors include seeking refuge from elevated stream temperatures, extreme flow conditions, and heavy predation, or seeking less densely populated areas with better opportunities for food and cover. For some juvenile fish, upstream migration is an important part of their life cycle.

Many studies indicate that a common strategy for over-wintering juvenile coho salmon is to migrate out of larger river systems into smaller streams during late-fall and early-winter storms to seek refuge from possibly higher flows and potentially higher turbidity levels in mainstem channels (Skeesick 1970; Cederholm and Scarlett 1981; Tripp and McCart 1983; Tschaplinski and Hartman 1983; Scarlett and Cederholm 1984; Sandercock 1991; Nickelson et al. 1992). There is evidence that coho salmon juveniles over-wintering in these areas have higher survival rates due to reduced water velocities in these microhabitats. Recent research conducted in coastal northern California suggests that juvenile salmonids migrate into smaller tributaries in the fall and winter to feed on eggs deposited by spawning adults as well as flesh of adult carcasses. Artificial impediments, such as road crossings or low flows, that restrict movement of juvenile coho salmon can reduce survival.

Numerous hydropower and water storage projects have been built that either block access to areas used historically by coho salmon or alter the hydrograph of downstream river reaches. NMFS (1995) identified a total of nine dams in California that currently have no fish passage facilities to allow coho salmon access to former spawning and rearing habitats. Blocked habitat

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constitutes approximately 9 to 11 percent of the historical range of each coho salmon ESU. There are five major dams within the California portion of the SONCC Coho ESU that currently block access to historical spawning and rearing areas of coho salmon. Combined, these blocked areas amount to approximately 11 percent of the freshwater and estuarine habitat in this region (Table 12). There are four major dams within the CCC Coho ESU that currently block access to historical spawning and rearing areas of coho salmon. Combined, these blocked areas amount to approximately nine percent of the freshwater and estuarine habitat in this region (Table 13). In addition to these, there are also five smaller impoundments on the mainstem Russian River, and approximately 500 licensed or permitted dams on its tributaries (SEC 1996).

Table 12. Major dams within the California portion of the Southern Oregon/ Northern California Coast Coho ESU, that block coho salmon from accessing historical spawning and rearing habitat (no passage facility available).

Name of Dam	Location	Upstream Habitat Blocked	Percent of Basin
Scott Dam	Eel River, approximately 169 miles (272 km) upstream from the Pacific Ocean, and forms Lake Pillsbury in Lake County, California	36 Miles (58km)	8% (Eel River Basin)
Matthews Dam	Mad River, approximately 79 miles (127 km) upstream from the Pacific Ocean, and forms Ruth Lake in Trinity County, California	2 Miles (3 km)	13% (Mad River Basin)
Lewiston Dam	Trinity River (tributary to the lower Klamath River), approximately 112 miles (179 km) upstream from the Pacific Ocean, and forms Lewiston Reservoir in Trinity County, California	109 Miles (175 km)	24%(Trinity Basin) 9% (Klamath Basin)
Dwinnel Dam	Shasta River (tributary to the upper Klamath River), approximately 214 miles (345 km) upstream from the Pacific Ocean, and forms Dwinnell Reservoir in Siskiyou County, California.	17 Miles (27 km)	17% (Shasta basin) 2% (Klamath basin)
Iron Gate Dam	Klamath River, approximately 190 miles (306 km) upstream from the Pacific Ocean, and forms Iron Gate Reservoir in Siskiyou County, California.	30 Miles (48 km)	8% (Klamath basin)

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Table 13. Major dams within the Central California Coast Coho ESU, that block coho salmon from accessing historical spawning and rearing habitat (no passage facility available).

Name of Dam	Location	Upstream Habitat Blocked	Percent of Basin
Peters Dam	Lagunitas Creek, approximately 14 miles (23 km) upstream from the Pacific Ocean, and forms Kent Lake in Marin County, California	8 Miles (13 km)	6%
Nicasio Dam	Nicasio Creek, (tributary to Lagunitas Creek), approximately 8 miles (13 km) upstream from the Pacific Ocean, and forms Nicasio Reservoir in Marin County, California.	5 Miles (8 km)	10%
Warm Springs Dam	Dry Creek (tributary to the Russian River), approximately 45 miles (72 km) upstream from the Pacific Ocean, and forms Sonoma Lake in Sonoma County, California.	50 Miles (80 km)	9%
Coyote Dam	Russian River, approximately 95 miles (153 km) upstream from the Pacific Ocean, and forms Lake Mendocino in Mendocino County, California.	36 Miles (58 km)	7%

Gravel Extraction

Introduction: As stated by Kondolf (1993), “The river can be likened to a conveyor belt, moving sediment eroded from the steep headwaters to ultimate deposition below sea level. By removing sediment from the active channel, instream gravel mining interrupts the continuity of this sediment transport and induces channel incision... .” The effects of gravel mining (instream and terrace) on aquatic resources is, therefore, very complex and cannot be adequately described unless the watershed as a whole is included in the study. This complex and interrelated nature of instream mining has made regulation of this activity extremely difficult. Kondolf (1995) asserts that state regulation of instream gravel mining “has been ineffective at preventing resource degradation, despite numerous permit requirements from various federal, state, and local agencies.” Until very recently, lead agencies (largely counties) that govern mining did not regulate instream mining at the watershed level. While impacts to rivers from instream mining have been documented in the past, the level of impact that instream mining has had on aquatic and riparian resources lessened in the mid-1990s with increased regulation.

Instream mining was originally controlled through FGC 1603 (Lake and Streambed Alteration Agreement), which was adopted in 1961. Instream mining is also regulated by Section 10 of the Rivers and Harbors Act and Sections 401 and 404 of the CWA (formerly known as the Federal Water Pollution Control Act of 1899). State regulation of mining has largely been accomplished under the California Environmental Quality Act (CEQA) of 1970 and the Surface Mine and Reclamation Act (SMARA) of 1975. However, specific state-wide standards that

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require the regulation of instream mining on a watershed basis were not adopted until 1991, and only apply to those operations permitted after 1991. Those standards state (Article 9 Section 3710 (c) & (d)) that:

“Extraction of sand and gravel from river channels shall be regulated to control channel degradation in order to prevent undermining of bridge supports, exposure of pipelines or other structures buried within the channel, loss of spawning habitat, lowering of ground water levels, destruction of riparian vegetation, and increased stream bank erosion (exceptions may be specified in the approved reclamation plan). Changes in channel elevations and bank erosion shall be evaluated annually using records of annual extraction quantities and bench-marked annual cross sections and/or sequential aerial photographs to determine appropriate extraction locations and rates... instream mining activities shall not cause fish to become entrapped in pools or in off-channel pits, nor shall they restrict spawning or migratory activities.”

At a more regional scale and recent time frame, regulation and consideration of cumulative impacts are being considered through local aggregate management plans such as Sonoma County Aggregate Resources Management Plan and Environmental Impact Report (Sonoma County Board of Supervisors 1994), Humboldt County’s Memorandum of Agreement and Programmatic Environmental Impact Report on Gravel Removal from the Lower Mad River (Humboldt County Board of Supervisors 1994) and the County of Humboldt Extraction Review Team, coupled with Lake and Streambed Alteration Agreements, and United States Army Corps of Engineers (USACE) Letters of Permission (LOP 96-1, 1996; LOP 96-2, 1997).

Potential impacts: Instream mining (the removal of sediment from the active channel) causes various impacts to salmonid habitat by interrupting sediment transport and often causing channel incision and degradation (Kondolf 1993). The classes of impacts that can result from instream mining include: direct mortality; noise disturbance; disruption of adult and juvenile migration and holding patterns; stranding of adults and juveniles; increases in water temperature and turbidity; degradation of juvenile rearing habitat; destruction or siltation of redds; increased channel instability and loss of natural channel geometry; bed coarsening; lowering of groundwater elevation; and loss of LWD and riparian vegetation (Humboldt County Public Works 1992; Kondolf 1993; Jager 1994; Halligan 1997). Terrace mining (the removal of aggregate from pits isolated from the active channel) has the potential to cause similar impacts to salmonids, when and if a flood causes channel capture by the gravel pits.

Direct impacts of gravel extraction: Direct impacts are those that are caused by the project and occur at the same time and place as the project. These include risk of direct mortality, impacts from noise, summer crossing construction, disruption of migration and holding patterns, and increased potential for stranding.

Direct mortality of salmonids may result from operations such as wet stream crossings by equipment, wetted channel dragline excavation, destruction of incubating salmonid eggs, and deposition of material into the rivers. The period for extraction operations coincides with the summer period of low flow, the end of the downstream salmonid migration run, beginning of the upstream spawning run, and low rainfall periods. Wet stream crossings are used only to facilitate the construction of summer bridges.

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Improper location, timing, and type of summer crossing installation may have direct impacts on salmonids in a number of ways. Improper location may reduce rearing habitat by filling in downstream pools. Installation of bridges in May or early June could possibly result in disturbance or burial of incubating or emerging salmonids. Construction of dry summer crossings could result in burial of fry and interference with upstream and downstream migration (Williams and Bjornn 1997).

Stranding is one of the risks that migrating salmonids face and can be exacerbated by human activities such as rapid flow reductions downstream of power-producing dams or gravel extraction activities that leave depressions and low spots on the gravel bars. Stranding primarily occurs when river stages rise and allow fish to move into newly inundated areas. As flows recede, fish can become trapped in pools or depressions found in overflow channels, isolated meander oxbows, around LWD, extraction bars, or other features. Unless flows increase or the depressions are fed by sub-surface flow, the trapped fish can succumb to high water temperatures or predators. Stranding of adult salmonids (excluding coho salmon) has been observed in the past on shallow riffles on the lower Eel River (Arcata Redi-Mix 1993). Halligan (1997) observed stranded adult chinook on shallow riffles in the Van Duzen River upstream of the Leland Rock operation during the summer of 1996. The Van Duzen fish were trapped by receding stormflows. Improper reclamation and regrading of gravel bars after extraction may result in the stranding of adult or juvenile salmonids. Fish may get trapped in shallow depressions as storm flows recede. Unless rescued in a short time, these fish will likely succumb to desiccation or predation.

Indirect impacts of gravel extraction: Indirect impacts are those that are caused by the project and are later in time or farther removed in distance from the project site. Potential indirect impacts that may occur from gravel mining operations include increased water temperatures, elevated turbidity, degradation of juvenile rearing habitat, reduction in spawning habitat, degradation of tributary mouths that can inhibit upstream migration, and loss of LWD that could reduce habitat diversity.

The effects on water temperature caused by gravel operations result from changes in channel morphology. Gravel skimming creates a less confined, wider channel. If the water level rises during the summer months, it could spread out over wide gravel bars instead of being confined in the relatively deep, narrow low-flow channel. The greater water surface area absorbs more incoming short-wave solar radiation, and the water temperature rises. Furthermore, the incoming summer solar radiation penetrates the relatively clear, shallow water and warms the gravel substrate. The gravel substrate releases long-wave radiation and helps maintain warm water temperatures into the evening hours. Warm temperatures can reduce fecundity, decrease egg survival, retard growth of fry and smolts, reduce rearing densities, increase susceptibility to disease, and decrease the ability of young salmon and trout to compete with other species for food and to avoid predation (Spence et al. 1996; McCullough 1999).

Coho salmon are susceptible to problems related to increased stream temperature because they usually spend a year in fresh water, thus are subject to temperature impacts associated with summer and early-fall. They are also less tolerant of warmer water temperatures than steelhead trout (Frissell 1992). In northern California, both Welsh et al. (2001) and Hines and Ambrose (1998) found that coho salmon juveniles were absent in stream reaches where the moving weekly

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average water temperature exceeded 62.2E F, or the moving weekly maximum temperature exceeded 64.9E F.

Conclusions: While instream gravel extraction has had direct, indirect, and cumulative impacts on salmonid in the recent past, no direct impacts to coho salmon have been documented under the current (post-1995) mining regulations as implemented through SMARA, local aggregate management plans, and USACE's Letters of Permission. However, many of the rivers within the petition area continue to feel the effects of years of channel degradation from the millions of tons of aggregate removed from the systems over time (Collins and Dune 1990). Therefore, indirect and cumulative impacts to coho salmon caused by current instream mining activities, such as increased braiding resulting in increased temperatures, have not been demonstrated conclusively.

Suction Dredging

Suction-dredge placer miners extract gold from the river gravels by sucking the gold-bearing gravels into the floating dredges, pumping the gravel-water mixture across a settling table where the gold concentrates by gravity, then discharging the gravel and water back into the river. An annual permit from the Department (under Title 14 CCR, section 228) and, in some circumstances, a Lake and Streambed Alteration Agreement is required to engage in this activity.

Dredges use high-pressure water pumps driven by gasoline-powered engines. The pump creates suction in a flexible intake pipe with a nozzle no greater than six inches in diameter. Suction dredges vacuum the streambed (which is composed of rock, gravel, and finer sediment) with water through the hose into the sluice box. Both the pump and the sluice box are usually mounted on a floating platform, often positioned over the work area by securing to trees or rocks with ropes or cables.

The portion of stream bottom dredged ranges from a few small excavations to the entire wetted area in a section of the stream. Larger suction dredges have the capacity to excavate as much as several cubic yards of gravel from the river bottom, depending on the type of streambed material and the skill of the operator.

Dredging activities in freshwater environments can have a variety of direct impacts on the environment, including impacts to aquatic and riparian organisms (Griffith and Andrews 1981; Thomas 1985; Harvey 1986) and channel stability. Impacts can also result from the potential release of hazardous constituents (such as mercury) to marine and terrestrial environments. However, there are no studies that document such dredging-related impacts on coho salmon or their habitat within the petitioned area.

Conclusion: Suction dredging, in accordance with Title 14 CCR, section 228, is allowed within the waters of the petitioned area. The restrictions currently imposed by regulations on this activity are designed to eliminate the potential for impacts to coho salmon by restricting suction dredging actions to locations and times when such activities will not impact the species.

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Streambed Alteration

In ecologically healthy watersheds, interactions between water flows, stream channels, and riparian vegetation produce habitat complexity and variety (Naiman et al. 1992). The complexity of these streams is used by a number of species at some point in their life cycle (Everest 1987). In forested areas, the LWD that falls into streams help create deep pools, trap sediment, stabilize stream banks, and produce varying water current (Bisson et al. 1987) which form additional habitat including side channel habitat or “flow shadows”. The resulting complexity allows the stream and its biological communities to adjust to natural disturbances such as fires, windstorms, and landslides, and creates a dynamic, productive ecosystem for coho salmon.

Streambed alteration activities can result in simplification of stream channels and a loss of habitat complexity (Bisson et al. 1987). Simplification effects include decreases in the range and variability of stream flow velocities and depths, and reductions in the amount of large wood and other stream structures. Activities in the stream channel can cause excess sediment to fill-in the pools, as well as eliminate the source of LWD that creates pools. Straightening of channels, or “channelization”, alters the geomorphology of the stream that creates channel complexity. Channelization that includes concreting the channel bottom permanently alters the substrate, and eliminates macroinvertebrate habitat, permanent in-stream vegetation, and natural substrate necessary for spawning.

Most of the coho salmon range in California consists of either pool-riffle or braided channels, or combinations of the two. Pool-riffle channels have an undulating bed which defines a rhythmic sequence of bars, pools, and riffles. Pools and riffles represent the topographic low and high points along the channel bed. Pools typically develop along one bank with an exposed bar along the opposite bank. The line of maximum depth (thalweg) commonly alternates from one bank to the other, crossing over at riffles.

Channel morphology adjusts to changing water and sediment discharges to maintain dynamic equilibrium. Often human activities induce changes by creating disequilibrium conditions which must then readjust to approach a new equilibrium. There are both direct and indirect changes. Direct changes include dam construction, water diversion, instream gravel mining, and channelization, while indirect changes include land use changes of many types. The time frame of recovery is dependent upon the sensitivity of the system to perturbation.

Gravel and cobble-sized sediment has tremendous ecological importance, as habitat for benthic macroinvertebrates and as spawning habitat for coho salmon. Sand and finer-grained sediment can degrade gravel and cobble habitats, especially when introduced to the channel at low flows, when they may accumulate on the bed. Most of the sediment transport occurs during floods, whether of short duration in rainfall-driven systems, or of longer duration but less intense snowmelt-driven systems. Sediment transported during high water flow episodes can move quickly through the system. When there is sediment deposition during low flows, sediment will remain in the channel substrate, thus impacting the gravel quality for coho salmon spawning and successful egg survival.

Loss of riparian habitat: Vegetation along streambanks exerts a strong control over bank stability, and thus has some influence on channel form. The role of riparian vegetation has,

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in general, been understated in part because of the additional complexity introduced by vegetative root patterns and problems in quantitatively defining the effects. The effects of vegetation are particularly evident in small channels where root masses from grasses may protect banks against even extreme flood events. In moderate-sized channels, the dense root mats formed by continuous bands of riparian vegetation such as alders, willows, and cottonwoods have been observed to maintain bank stability in 20-50-year events depending upon flow orientation relative to the bank. LWD in small, forested channels has been shown to be responsible for 50-90% of the pool formation and 60% of the total fall in stream elevation by creating “organic” riffles.

The community health of stream-riparian ecosystems requires periodic disturbance and renewal to create a wide diversity of habitats based on variable age and species composition. Natural mortality agents include dessication, inundation, erosion (undermining and damage by bedload), and breakage through debris impacts during high flows.

Loss of pool habitat: Activities that increase sediment yield from the watershed and deposition in downstream reaches can impact pool habitat as well as gravel quality. In many systems, pool habitat has been lost to filling by fine sediment, reducing the abundance of pools. The *pool-riffle ratio* provides one measure of this. Activities that decrease in-stream LWD or recruitment of LWD, affect the geomorphology of streams by decreasing the structures that help the formations of pools.

Navigational improvement activities: Most of the long-term damage to the aquatic environment from navigational improvement activities has already occurred in the form of habitat alterations. North and central coast streams and rivers in their natural state are littered with LWD, and their complex channels consist of oxbows, multiple channels, and small impoundments that create the complex habitat required for coho salmon reproduction and survival. Navigation, on the other hand, requires deep, straight channels, free of snags that could harm boat hulls and propellers.

Roads: Road building is a component of many different land-use activities, and the total amount of road surface area in California is substantial. Stream and riparian habitats are routinely damaged while building roads because many roads wind their way through stream corridors. In the process, many streams are channelized to prevent erosion of stream banks that have roads built on top. Roads contribute to increased runoff and increased delivery of contaminants and inorganic sediment to streams and rivers. Compacted gravel or dirt and paved asphaltic roads are nearly all impervious surfaces that allow no infiltration. Watershed-scale changes in permeability has been found to compromise flow regimes, essential physical characteristics, and water chemistry in lower-order spawning and rearing streams in the Pacific Northwest (May 1997).

Roads can also deliver large volumes of inorganic sediment to streams and rivers, especially from poorly maintained rural and forest roads. Mass-wasting or the delivery of large volumes of soil to the stream through land slides is a symptom of poorly built roads, or well-built roads on unstable geology. One large factor contributing to mass-wasting is when two or more channels upslope of a road are combined through one culvert under the road and directed into one of the stream channels downslope of the road. This is usually done with smaller streams, but doubling or tripling the flow through a channel will inevitably cause mass erosion of the channel

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that can take large sections of road with it, delivering enormous amount of sediment to a stream channel.

Water Quality

Under Section 303(d) of the 1972 CWA, states, territories and authorized tribes are required to develop lists of impaired waters that do not meet water quality standards, even after parties responsible for point sources of pollution have installed the minimum required levels of pollution control technology. The law requires that these jurisdictions establish priority rankings for water on the lists and develop action plans, including TMDL plans to improve water quality.

TMDLs in California are developed either by RWQCBs or by the USEPA. TMDLs developed by RWQCBs are designed as Basin Plan amendments and include implementation provisions. TMDLs developed by USEPA typically contain the total load and load allocations required by Section 303(d), but do not contain comprehensive implementation provisions. An implementation plan is required in order for the RWQCBs to incorporate any TMDL into their Basin Plans. In developing implementation programs for TMDLs established by the USEPA, it has often become necessary for the RWQCBs to justify or change the USEPA numbers. This process has resulted in the RWQCBs adopting TMDLs that are different from those established by the USEPA. Within the range of coho salmon, there are 74 water bodies that are on the section 303(d) list of impaired water bodies (Table 14).

Water pollution originates from point sources and non-point sources as listed in Table 14. The combined input and at times unknown origin of nutrients, biocides, metals and metalloids make it difficult to pinpoint specific pollutants to specific and direct effects on coho salmon. Mixed compounds have different effects on the biological community of a stream than would any single compound. In addition, effects vary with habitat alteration, temperature and the concentration of dissolved materials in the surface waters (Brown and Sadler 1989). Water quality within the coho salmon range is affected by industrial discharges, agricultural discharges, mineral mining wastes, municipal wastewater discharge, road surface discharge, and urban stormwater discharge.

The following discussion on water quality impacts generally applies to salmonids. These water quality impacts have not directly been shown to impact coho salmon in California.

Industrial discharges: There are many types of industry in California, primarily centered around urban areas. The types of discharges that industries produce are either chemical or organic. There are many industrial producers of chemical products and their discharge can contain several of the toxic pollutants listed in Table 14. Metal discharges such as cadmium, chromium, copper, iron, lead, mercury, nickel and silver are toxic to fish at low concentrations (USEPA 1986). McDonald et al. (1989) report a particular metal may be toxic to an aquatic organism because of its surface activity, in addition to any internal effects it might have. This difference in effect is due to the fact that aquatic organisms have more delicate external surfaces (in terms of structure and physiology) than the exposed surfaces of terrestrial animals. Amongst the ranked metals, copper was found to be the most toxic metal to trout, with nickel being the least toxic. Metals can also have dramatic effects in early life-stages of fish, both upon body calcium content and upon skeletal mineralization.

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Table 14. List of Clean Water Act Section 303(d) impaired water bodies within the range of coho salmon in California.

WATER BODIES AND AREA AFFECTED	STRESSOR	SOURCE OF POLLUTION ^j
SAN FRANCISCO BAY		
Carquinez Strait, 6560 Acres; Richardson Bay, 2560 Acres	Chlordane; Copper; DDT; PCBs; PCBs (dioxin-like); Diazinon ; Dieldrin; Dioxin compounds; Exotic species; Mercury; Furan compounds; Nickel; Selenium; High coliform count	1, 5, 6, 7, 20, 26, 27, 28, 34, 38, 45
San Francisco Bay 172,100 Acres	Chlordane ; Copper; DDT; Diazinon ; Dieldrin; Dioxin compounds; Exotic Species; Furan compounds; Mercury; Nickel; PCBs; PCBs (dioxin-like); Selenium; High coliform count	1, 5, 6, 7, 20, 26, 27, 28, 34, 36, 38, 47
San Pablo Bay, 71,300 Acres; Suisun Bay, 25,000 Acres; Suisun Marsh Wetlands, 57,000 Acres; Suisun Slough, 10 Miles	Chlordane ; Copper; DDT; Diazinon ; Dieldrin; Dioxin compounds; Exotic Species; Furan compounds; Mercury; Nickel; PCBs; PCBs; (dioxin-like); Selenium; High Coliform Count; Metals	1, 5, 6, 7, 15, 20, 26, 27, 28, 34, 36, 38, 45
Tomales Bay; Calero Res.; Guadalupe Res.; Lake Herman; Merritt Lake; Alameda Cr.; Alamitos Cr.; Arroyo Corte Madera Delpresidio; Arroyo De La Laguna; Arroyo Del Valle; Arroyo Hondo; Butano Cr.; Calabazas Cr.; Corte Madera Cr.; Coyote Cr. (Marin and Santa Clara Cos); Gallinas Cr.; Guadalupe Cr.; Lagunitas Cr.; Laurel Cr.; Ledge wood Cr.; Los Gatos Cr.; Matadero Cr.; Miller Cr.; Mt. Diablo Cr.; Napa R.; Novato Cr.; Permanente Cr.; Pescadero Cr.; Petaluma R.; Pine Cr.; Pinole Cr.; Rodeo Cr.; San Antonio Cr.; San Felipe Cr.; San Francisquito Cr.; San Gregorio Cr.; San Leandro Cr.; San Lorenzo Cr.; San Mateo Cr.; San Pablo Cr.; San Rafael Cr.; Saratoga Cr.; Sonoma Cr.; Stevens Cr.; Walker Cr.; Walnut Cr.; Wildcat Cr. (Total: 8520 Acres and 759 Miles)	Metals; Nutrients; Pathogens; Sedimentation/ Siltation; Mercury, Floating material; Org. enrichment/ Low D O; Diazinon; Pathogens; Salinity	1, 4b, 10, 15, 25, 28, 38, 42, 44, 45
NORTH COAST		
Albion River, 14 Miles	Sediment	28, 39
Eel River Delta, 6350 Acres	Sediment; Temperature	28, 31, 39
Eel River, 2508 Miles	Sediment; Erosion; Temperature	13, 16, 23, 28, 31, 32, 33, 34, 36, 39, 41
Elk River, 88 Miles	Sediment	39
Freshwater Creek. 73 Miles	Sediment	13, 16, 23, 28, 33, 34, 39
Garcia River, 39 Miles	Sediment; Temperature	13, 16, 23, 28, 32, 33, 34, 35, 39, 41

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Table 14, continued

WATER BODIES AND AREA AFFECTED	STRESSOR	SOURCE OF POLLUTION
Gualala River, 35 Miles	Sediment	13, 16, 22, 23, 28, 33, 34, 39, 20
Klamath River, 190 Miles	Nutrients, Org. enrichment/Low D. O.; Temperature	3, 11, 15, 17, 21, 26, 28
Mad River, 90 Miles	Sediment	28, 36, 39
Mattole River, 56 Miles	Sediment; Temperature	13, 17, 28, 31, 32, 35, 39, 40
Navarro River, 25 Miles	Sediment; Temperature	1,3, 8, 12, 13, 15, 16, 17, 21, 22, 23, 28, 31,32, 33, 34, 35, 36, 37, 39, 40, 41, 42
Noyo River, 35 Miles	Sediment	28, 39
Redwood Creek, 63 Miles	Sediment	28, 31, 39
Russian River, 105 Miles	Sediment	4a, 6, 8, 12, 15, 16, 17, 19, 22, 23, 28, 32, 33, 34, 35, 37, 39, 40, 41, 43, 45
Scott River, 68Miles	Sediment; Temperature	3, 12, 21, 25, 28, 30, 32, 36, 39, 41, 46
Shasta River, 52 Miles	Org. enrich/ Low D. O. Temperature	2, 3, 12, 15, 17, 28, 32, 35, 46
Stemple Creek, 17 Miles	Nutrients	28
Ten Mile River, 10 Miles	Sediment	28, 39
Trinity River, 250 Miles	Sediment; Temperature	17, 25, 28, 31, 32, 35, 36, 29, 41, 46
Van Duzen River, 63 Miles	Sediment	13, 28, 31, 39

^j 1- Agriculture; 2 - Agriculture- irrigation tailwater; 3 - Agricultural Return Flows; 4a - Agriculture- storm runoff; 4b - Animal Operations; 5 - Atmospheric Deposition; 6 - Ballast Water; 7 - boat Discharges/vessel wastes; 8 - Bridge Construction; 9 - Channel modification,channelization; 10 - Construction/ Land Development; 11 - Dam Construction and Operation; 12 - Drainage/ Filling Of Wetlands; 13 - Erosion/ Siltation; 14 - Filling Of Wetlands; 15 - Flow Regulation/ Modification; 16 - Harvesting; 17 - Habitat Modification; 18 - Highway/ Road/ Construction; 19 - Hydromodification; 20 - industrial Point source; 21 - Irrigated Crop Production; 22 - Land Development; 23 - Logging Road Construction/ Maintenance; 24 - Manure Lagoons; 25 - Mine Tailings; 26 - Municipal Point Source; 27 - Natural Sources; 28 - Nonpoint Source; 29 - Other Urban Runoff; 30 - Pasture Land; 31 - Range Land; 32 - Removal of Riparian Vegetation; 33 - Residue Management; 34 - Restoration; 35 - Riparian Grazing; 36 - Resource Extraction; 37 - Road Construction; 38 - septage disposal; 39 - Silviculture; 40 - Specialty Crop Production; 41 - Streambank Modification/ Destabilization; 42 - Surface mining; 43 - Upland Grazing; 44 - Upstream Impoundment; 45 - Urban Runoff/storm sewers; 46 - Water Diversions; 47 - Water (groundwater), domestic use

Treated industrial waste can cause chronic effects when the combination of substances discharged causes low-level toxicity or when the aquatic community in the receiving waters are already stressed from environmental conditions or other land-use activities (Dickson et al. 1987) Excessive discharges of chemical wastes result in acute toxicity and fish kills. Excessive organic waste discharges cause high biochemical oxygen demand (BOD) which causes fish kills. Industrial waste is often warmer than the receiving waters and therefore high inputs elevate ambient water temperatures.

Agricultural discharges: Grier et al. (1994) reviewed a lengthy list of pesticides which are known to disable coho salmon behaviorally or interfere with their reproductive fitness in some way. Neurotoxic pesticides are known to contaminate surface waters that provide habitat

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for salmonids, including some listed for protection under the ESA (Sholtz et al. 2000). Despite their widespread use, the impacts of these pesticides on the neurological health of wild salmon are not well understood. Of particular concern are the organophosphate and carbamate insecticides that block synaptic transmission. Scholz et al. (2000) assessed the effects of diazinon, an organophosphate insecticide, on anti-predator responses and homing behavior in chinook salmon. Nominal exposure concentrations (0.1, 1.0, and 10.0 FgL⁻¹) were chosen to emulate diazinon pulses in the natural environment. In the anti-predator study, diazinon had no effect on swimming behavior or visually-guided food capture. However, the pesticide significantly inhibited olfactory-mediated alarm responses at concentrations as low as 1.0 FgL⁻¹. Similarly, homing behavior was impaired at 10.0 FgL⁻¹. Results suggest that olfactory-mediated behaviors are sensitive to anticholinesterase neurotoxicity in salmonids and that short-term, sublethal exposures to these insecticides may cause significant behavioral deficits. Such deficits may have negative consequences for survival and reproductive success in these fish (Sholtz et al. 2000).

Mineral mining wastes: California's historical mining industry was developed in a time of less-sophisticated mining methods and before modern environmental regulations. Abandoned mines and mine wastes resulting from the historical extraction of gold, silver, and mercury (among other metals) continues to plague California. An issue associated with such legacy problems within the State is metal-loading. Acid rock drainage can occur when sulfide minerals are exposed to air and water as a result of the mining activity. A chemical and biological reaction takes place resulting in the creation of sulfuric acid, which dissolves metals and which, in high concentrations, can be very harmful to aquatic life. It is the metal-loading that causes a greater environmental concern than the acidity.

The largest numbers of acid-producing abandoned mines are outside the range of coho salmon in the copper/gold belt of the Sierra Nevada, and the largest and most famous acid-producing mine is Iron Mountain Mine near Redding. Formations that contain gold and silver and are also often rich in sulfides, do occur within the Klamath/Trinity River watershed. Within the range of coho salmon, exclusive of areas draining into San Francisco Bay, only four mines are on the State's List of Mines with Potentially Significant Environmental Hazards (California Department of Conservation 2000): Grey Eagle Tailings, Buzzard Hill, and Siskon (all in Siskiyou County); and Altoona (Trinity County). Acid rock drainage has been documented to some extent at these four mines. Water quality impacts to Indian Creek (a tributary to the Klamath River) by the Grey Eagle Tailings caused a USEPA clean-up in 1998. Coho salmon had been detected in Indian Creek in the 1970s and 1980s; however, they were not detected in the 2001 surveys.

Mercury contamination by abandoned mines occurs within the mercury mining belt of the Coast Ranges, and within the large placer and hydraulic mines of the Sierra Nevada and Klamath Ranges. The mercury was mined from the Coast Ranges and then tons of elemental mercury (a potent neurotoxin) were used to extract the gold in placer and hydraulic mining operations. Within the watersheds that drain into San Francisco Bay, metal-loading and mercury impacts are known at many small mines in Napa, Santa Clara, and Alameda counties, with one of the largest mercury mines, the New Almaden, draining into the South Bay. While mercury impacts to the aquatic environment have been documented within the Bay, there are no data to connect impacts to the aquatic environment from these mines, to direct impacts on coho salmon.

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The mercury that was used to recover the gold (and silver) from the large placer and hydraulic mines was lost to the environment and remains within the rivers and streams of the state, especially in the Sierra Nevada. Such large-scale placer and hydraulic mines occurred within the Scott, Salmon, Trinity, and Lower Klamath River watersheds as well. The total amount of mercury lost to the environment from all these operations may have been 3-8 million pounds or more (Churchill 1999). The bio-available form of mercury, methylmercury, has been detected in appreciable quantities in Sierra Nevada rivers, streams, and lakes; however, limited sampling by the U.S. Geologic Survey has yet to detect significant methylmercury within the drainages associated with the historical Klamath-Trinity hydraulic mines (Charlie Alpers pers. comm.).

While localized effects from metal-loading on fish and invertebrates have been documented in other areas of the U.S. (Rand and Petrocelli 1985) and within the Sierra Nevada watersheds (Charlie Alpers, pers. comm.), within the petitioned area, there are no data from the watersheds associated with the above referenced mines documenting direct or indirect impacts on coho salmon.

Municipal wastewater discharge: The most significant alteration that municipal wastewater has on stream hydrology is to increase the base flow of streams and small rivers with treated wastewater discharges. These discharges can result in increased algae production and eutrophication, which in turn alters the chemistry and ecology of the stream.

The primary components of municipal wastewater are nutrients and dissolved and suspended organic matter. Most of the nutrients are discharged as phosphorus and nitrogen in the form of NH_3 and NO_3 . Acute effects to aquatic organisms usually occur when there is an accidental spill of chlorine, or when the system becomes overloaded and too much of the nutrients are in the form of ammonia. However, excessive plant growth stimulated by nutrients and excessive suspended organic matter can cause occasional high BOD and resulting fish mortality. Chemical contaminants from household use, or when industrial discharge is routed through a municipal wastewater treatment plant, can cause occasional acute and chronic effects to aquatic organisms in the receiving waters. Municipal wastewater is often warmer than the receiving waters and therefore can elevate ambient water temperature. The discharge may also have the ability to change the DO, pH, or other ambient water quality parameters.

Road surface discharge: The building of roads is a component of many different land-use activities and the total amount of road surface area in California is substantial. Because many roads wind their way through stream corridors, streams and riparian habitats are routinely damaged by delivery of contaminants and inorganic sediment to streams and rivers. Compacted gravel or dirt and paved asphaltic roads are nearly all impervious surfaces or land surfaces that allow no infiltration, and where virtually all the rain water is runoff. Rain water, especially during the first few storms of the season, carries with it the oil, fuel, grease, and other chemicals that have accumulated on the road surfaces during the dry season, as well as the herbicides used to maintain roadside areas.

Urban stormwater discharge: One of the major issues regarding the effects of urbanization on water quality is the discharge of storm water, and this is one of the leading environmental problems in the United States (USEPA 1983). Storm water in urban areas is the combination of runoff from all impervious surfaces including roads, parking lots, and other

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surfaces that do not have vegetation growing on them. Storm water contains contaminants that fall on road and parking lot surfaces and house and lawn chemicals that are used by homeowners.

In addition to water contamination, storm water can cause hydrologic impacts and sedimentation problems. Concentration of runoff into storm drains will cause increased episodic flow events with higher peak flows. Natural stream channels receiving storm water have higher erosion and sedimentation in downstream reaches of the stream.

Agricultural Impacts

Historically, agriculture has used lands near bodies of water such as streams, ponds, or lakes. These lands were used for both animal husbandry and for a variety of crops, usually requiring that they be cleared of most existing vegetation (Booth 1991). Due to their proximity to water, riparian habitat is lost through these direct conversions to agriculture (Terrell and Perfetti 1989), and crops are often planted close to the river's edge. For example, the majority of vineyards in the Russian River basin are located within 300 feet of the riparian zone (CDFG 2001c). Even if a riparian buffer was originally established, it may later be eliminated in order to remove habitat that could harbor agricultural pests. An example of this is the agricultural practice in Sonoma and Mendocino counties of removing riparian vegetation bordering vineyards to decrease host plants for the xylem feeding insects that carry Pierce's disease (CDFG 2001c).

Agricultural practices affect aquatic and riparian areas through non-point source pollution, since these areas eventually receive sediments, fertilizers, pesticides, and wastes from associated agricultural lands. Sediment is the most common type of non-point source pollution from agricultural lands (Knutson and Naef 1997). According to Terrell and Perfetti (1989) erosion of crop lands accounts for 40 to 50 percent of the sediment in the United States waterways. Storm runoff erodes the topsoil from open agricultural areas, and irrigation water from standard agricultural practices also carries significant amounts of sediment to the stream environment. According to Terrell and Perfetti (1989), two types of irrigation systems, sheet flow and rill, cause the greatest amount of surface erosion, while drip irrigation and piped laterals produce the least. Irrigation requires water that is drawn from the stream, lake, pond, or from the ground. Pumping from the water table reduces its level, decreasing flow to and in the river. The ability for a stream to diminish the effects of irrigation waste discharged into a stream decreases proportionally with the flow.

Small coastal streams often rely on springs to maintain flows throughout the summer months, however the aquifers that supply these streams are often utilized for irrigation. Many streams that once flowed year-round no longer do so, because of recent increases in hillside agricultural land conversion. The conversion of uplands from forest or grasslands to steep agricultural steppes, increases the erosion potential and ground water use (CDFG 2001c). Often these converted agricultural lands are for vineyards. In February 2000, Sonoma County adopted a vineyard ordinance to control sedimentation caused by vineyard erosion (Merenlender et al. 2000). The ordinance identified three levels of vineyards and seven types of "highly erosive" soils and provides corresponding requirements (CDFG 2001c).

Animal wastes carried by runoff can contaminate water sources through oxygen-depleting organic matter (Knutson and Naef 1997). Runoff from concentrated fecal sources can change

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water quality, causing lethal conditions for fish. As the BOD increases, DO decreases, and ammonia is released causing changes that are stressful to fish.

Grazing can affect riparian characteristics and associated aquatic systems, such as vegetative cover, soil stability, bank and channel structure, instream structure, and water quality and quantity. Behnke and Zarn (1976) and Armour et al. (1991) indicated that overgrazing is one of the major contributing factors for Pacific Northwest salmon declines. Trampling may compress or compact soils, decreasing water infiltration and increasing runoff. However, light trampling can break up surface soils that have become impervious, and allow for greater water absorption; however, this also makes the soil more susceptible to erosion (Spence et al. 1996). According to Knutson and Naef (1997), some of the ways that poor grazing practices can impact fish and wildlife include:

- reducing or eliminating regeneration of woody vegetation;
- changing plant species composition in favor of more xeric species (trees, willows, and sedges replaced by brush and bare soil);
- reducing overall riparian vegetation;
- loss of protective vegetation that increases bank and instream deformation and stabilization;
- trampling and soil compaction;
- increasing stream bank erosion, which causes stream channel widening, shallowing, trenching, or braiding;
- reduced ability for riparian areas to trap and filter sediments and pollutants;
- increasing stream temperatures due to loss of cover;
- increasing the magnitude of high and low flows;
- lowering the water table, and associated loss of riparian vegetation; and
- loss of nutrient inputs, especially invertebrate food sources, to stream.

Impacts such as these can be observed, to varying degrees, throughout the coho salmon California range.

Urbanization

Humans have traditionally settled near sources of water, such as streams, lakes, and bays. Though the effects of timber, livestock, and agriculture can be destructive, there is usually the chance for recovery of the landscape. In urban areas, recovery is unlikely, because once the natural vegetation is gone and the stream and riparian habitats are modified, the changes are usually permanent (Booth 1991; Spence et al. 1996). Booth (1991) indicated that urban watersheds could increase maximum discharge, associated with storm and flood events, as much as five times over natural stream conditions. Areas within the range of coho salmon where large-scale urban development has taken place include Arcata-Eureka, Fortuna, Willits, Ukiah, Santa Rosa, Marin County, and the San Francisco Bay Area.

Fishing and Illegal Harvest

Retention of coho salmon has been prohibited in ocean commercial fisheries south of Cape Falcon, Oregon beginning with the 1993 season. From Cape Falcon to Horse Mountain, California, coho salmon retention has been prohibited in ocean recreational fisheries since the 1994 season, and

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starting May 1995, the prohibition was extended to include sport fisheries south of Horse Mountain. California's inland waters have explicitly been closed by regulation to coho salmon retention since 1998.

Prior to these restrictions, fisheries for coho salmon occurred along the entire west coast of North America as far south as central California. Most coho salmon originating from Pacific coast states recruit to the fisheries after one year in fresh water and about 16 months at sea. Existing fisheries take place in coastal adult migration corridors, near the mouths of rivers, and in freshwater migration areas, largely targeting fish returning to streams with hatcheries. Trolling (hook-and-line) is the primary gear type used in commercial fisheries; however, gill nets and purse seines are used in some nearshore or in-river fisheries. Sport catches of coho salmon are typically taken by hook-and-line.

Coho salmon are also taken incidentally in fisheries directed toward other salmon species. When regulations prohibit the retention of coho salmon, the majority of released fish survive the hooking encounter. However, if large enough numbers are hooked, substantial mortality can be incurred. Substantial coho salmon bycatch can lead to restrictions on these non-directed fisheries.

Ocean Harvest

Coho and chinook salmon have historically constituted the major species taken in the Pacific coast commercial and recreational salmon fisheries. The fisheries off California had been directed toward and harvested primarily chinook, in contrast to those off Washington and Oregon which have largely targeted coho salmon.

Ocean commercial harvest of coho salmon in California peaked during the period 1961 through 1980, when five-year averages ranged from 150,280 to 361,660 fish. Since 1986, total harvest had not exceeded 83,000 fish annually (Table 15). Approximately 80% of the catch was landed in ports north of Point Arena. Since 1993, when non-retention was instituted, no known landings of coho salmon have been seen (PFMC 2001a).

Harvest of coho salmon was generally lower in California's ocean recreational fishery than in the commercial fishery. Since 1962, when reliable records were first taken, total harvest has not exceeded 70,000 fish (Table 16). As in the commercial fishery, most (88%) of the recreational coho salmon catch was landed from Point Arena northward. The salmon fishery in this area is characterized by large numbers of private boats and few charter boats, while in the fishery south of Pt. Arena, party boat operation is more prevalent. Non-retention of coho salmon, starting in 1994, has greatly reduced the harvest, although there continued to be a small number (less than 1000) of fish incidentally caught and illegally landed (PFMC 2001a).

Coho salmon harvested off California probably consisted of a mixture of fish originating from both Oregon and California streams and hatcheries. Federal regulation of the ocean fisheries recognizes this, and manages both states' stocks as an aggregate. One of the management tools used to determine salmon harvest rates is the recovery of coded-wire tagged salmon through statistically appropriate, randomized sampling programs. Tagged fish recoveries have been used to estimate the total occurrences of a particular release group in all of the fish caught. However, it has not been possible to determine the composition of California's contribution to the coho salmon ocean harvest from coded-wire tagged recoveries of landed fish because of inadequate and inconsistent tagging rates among its hatchery- and naturally-produced fish.

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Table 15. Harvest of coho salmon in the California commercial fishery for major ports of landings, 1952 - 2000.^k

Year(s)	Average number or number of fish landed					
	Crescent City	Eureka	Fort Bragg	San Francisco	Monterey	TOTAL
1952-1955	33,075	23,675	18,950	2,300	500	78,500
1956-1960	12,240	9,740	15,900	4,960	1,000	43,840
1961-1965	40,720	47,060	40,060	18,780	3,660	150,280
1966-1970	86,400	99,760	70,840	55,960	6,760	319,720
1971-1975	84,020	133,940	100,420	35,500	7,780	361,660
1976-1980	72,120	89,920	51,020	20,760	9,400	243,220
1981-1985	16,100	18,900	14,600	7,700	1,400	58,700
1986	4,800	4,300	20,800	5,100	1,300	36,300
1987	5,800	10,800	25,900	1,200	100	43,800
1988	2,800	10,100	30,900	6,700	400	50,900
1989	5,800	3,400	25,800	6,500	500	42,000
1990	-	1,200	26,600	27,400	5,700	60,900
1991	-	3,000	4,500	53,300	21,400	82,200
1992	-	-	-	400	2,100	2,500
1993	-	-	-	-	-	-
1994	-	-	-	-	-	-
1995	-	-	-	-	-	-
1996	-	-	-	-	-	-
1997	-	-	-	-	-	-
1998	-	-	-	-	-	-
1999	-	-	-	-	-	-
2000	-	-	-	-	-	-
TOTAL	363,875	455,795	446,290	246,560	62,000	1,574,520
% of harvest	23.1%	28.9%	28.3%	15.7%	3.9%	

^k Retention of coho salmon has been prohibited since in 1993.

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Table 16. Harvest of coho salmon in the California recreational fishery for major ports of landing, 1962 - 2000.¹

Year(s)	Average number or number of fish landed					TOTAL
	Crescent City	Eureka	Fort Bragg	San Francisco	Monterey	
1962-1965	1,725	13,250	6,425	3,163	1,850	26,413
1966-1970	2,600	13,760	5,460	8,820	2,520	33,160
1971-1975	6,880	22,200	9,020	8,760	1,450	48,310
1976-1980	10,640	13,080	2,440	3,620	150	29,930
1981-1985	7,400	10,400	900	1,100	100	19,900
1986	8,100	8,600	1,600	400	50	18,750
1987	14,900	29,800	2,500	100	50	47,350
1988	12,200	18,300	3,200	400	200	34,300
1989	18,500	26,400	3,700	900	200	49,700
1990	15,500	24,600	4,500	5,800	1,200	51,600
1991	18,300	21,800	18,600	7,700	2,900	69,300
1992	2,800	3,600	3,300	1,600	200	11,500
1993	6,700	7,600	12,300	3,000	200	29,800
1994	100	--- ^m	200	200	--- ^m	500
1995	100	200	500	200	--- ^m	1,000
1996	100	200	300	100	--- ⁿ	700
1997	100	100	100	200	--- ^m	500
1998	--- ^m	--- ^m	--- ^m	--- ^m	--- ^m	100
1999	--- ^m	100	200	300	--- ⁿ	600
2000	100	100	100	100	100	500
TOTAL	126,745	214,090	75,345	46,463	11,170	473,913
% of Total	26.7%	45.2%	15.9%	9.8%	2.4%	

¹ Retention of coho salmon had been prohibited north of Horse Mountain, CA starting in 1994, and in 1995 was extended to south of Horse Mountain. Numbers shown for those years and subsequent represent estimated incidentally taken and illegally landed fish.

^m Less than 50 fish.

ⁿ No data

The impact that commercial and recreational ocean fishing has had on the long-term decline of coho salmon populations is not clear. There are few historical or recent records to indicate that curtailment of fishing has increased coho salmon abundance. While curtailment of fishing seasons has been thought to have reduced harvest-related mortality rates on Oregon coastal coho salmon populations substantially, there has been no evidence of a corresponding increase in coho salmon spawner escapement there.

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Inland Harvest

Sport fisheries for coho salmon in northern California coastal streams were not extensive and for the most part, were concentrated primarily in the estuarine areas. The fishery has not been monitored in most of the tributaries, and since 1977 the most consistent coho salmon harvest data is available only for the Klamath-Trinity river system (Pacific States Marine Fisheries Commission 2001). Highest annual catch in these rivers was estimated at about 3,600 coho salmon in 1987, but for the majority of the years less than 500 fish were caught (Table 17).

The Klamath basin's native American tribes (Yurok, Hoopa, and Karuk) also harvested coho salmon, and currently constitute the only existing sanctioned fishery directed toward the species. Both the Yurok and Hoopa Valley tribes have federally recognized fishery rights in the basin, and tribal subsistence, ceremonial, and minor commercial fisheries are prosecuted under regulatory authority of each respective tribe. Each tribe determines the level of fishing opportunity that will be provided its tribal members based on estimates of preseason abundance. Data for this review is only available for the Yurok tribe's harvest, resulting from subsistence and ceremonial fisheries within the tribe's reservation on the lower Klamath River (Weitchpec downstream to the ocean); these fisheries have only been monitored since 1992. Harvest has ranged from 27 to 1,168 fish caught annually (Table 17), and based on estimates of upstream escapement (in-river spawners and hatchery returns), is thought to be an average 4.4% harvest rate for the period (Dave Hillemeier pers. comm.).

Illegal Harvest

Illegal harvest can have an impact on populations of fishes in certain areas, although this depends on intensity, frequency and species of fish taken. The Wildlife Protection staff of the Department was queried regarding illegal harvest of coho salmon in California. Their responses indicated that illegal harvest of both juvenile and adult coho salmon does occur, although most of the illegal take is due to anglers mistaking coho salmon for some other species. Most of the violations involving the illegal take of adult coho salmon occur in the offshore sport fishery. Illegal harvest in inland waters is mostly opportunistic, meaning poachers will spear, net, gaff or snag whatever salmonid that happens to be in the stream (Tom Belt pers. comm.).

Overall, Department enforcement staff did not believe that many coho salmon are illegally harvested, and do not believe this is a significant impact on California populations. The major reasons cited as to why few coho salmon are illegally harvested in California are that coho salmon migrate during high flows and are not concentrated. The few fish that are retained by sport anglers are most often due to misidentification.

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Table 17. Estimated harvest of coho salmon in the Klamath/Trinity River system..

Year	Number of fish caught		
	Recreational sport fishery		Yurok tribal fisheries °
	Trinity River	Klamath River	
1977	149	30	---
1978	0	13	---
1979	827	484	---
1980	0	---	---
1981	966	---	---
1982	476	---	---
1983	1,674	34	---
1984	182	---	---
1985	763	92	---
1986	750	60	---
1987	3,368	233	---
1988	1,961	489	---
1989	300	273	---
1990	47	54	---
1991	109	28	---
1992	24	44	122
1993	64	9	1,168
1994	0	4	27
1995	294	49	830
1996	248	160	953
1997	42	0	78
1998	0	---	181
1999	98	---	235
2000	---	---	91

° Consists of subsistence and ceremonial harvest, which has only been monitored since 1992.